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Habitat use and movement by brook trout (Salvelinus fontinalis) and rainbow trout (Oncorhynchus mykiss) in three tributaries of Nipigon Bay, Lake Superior

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Habitat Use and Movement by Brook Trout (*Salvelinus fontinalis*) and Rainbow Trout (*Oncorhynchus mykiss*) in Three Tributaries of Nipigon Bay, Lake Superior

by

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ABSTRACT

Knowledge of the physical habitat features of streams that are used by brook trout (*Salvelinus fontinalis*) and rainbow trout (*Oncorhynchus mykiss*) is an essential component of insuring that the habitat is protected from potential impacts of land-use. In three small streams along the north shore of Lake Superior I examined the characteristics of the stream habitat and length of stream generally occupied by the brook trout and rainbow trout during the summer, including brook trout in the presence and absence of rainbow trout in the same stream. Electrofishing, passive integrated transponder (PIT) tagging, minnow trapping, two-way weirs, and habitat manipulations were techniques utilized in this study. Catch rates and fish biomass were examined in relation to associated characteristics of the stream habitat. The length of stream generally occupied was extrapolated from the capture locations of PIT-tagged brook trout and rainbow trout. Two-way weirs were placed at the top and bottom of a reach within a section of one of the streams to monitor immigration and emigration of brook trout and rainbow trout. Brook trout catch-per-unit-effort (CUE) using baited minnow traps was more variable when traps were placed near low to high amounts of structure in pools compared to areas with no structure. Whereas, rainbow trout CUE using baited minnow traps was low for all categories, from no stucture to high amounts of structure. Brook trout biomass for entire pools was found to be significantly correlated with canopy closure over pools in the absence of rainbow trout ($r^2 = 0.056$, $p = 0.037$) and in the presence of rainbow trout ($r^2 = 0.059$, $p = 0.044$). However, no significant correlations were found between rainbow trout biomass for entire pools and various physical features that were measured within pools. The majority of re-captured brook trout (20/23) and rainbow trout (13/15) were less than two pools from the pool in which they were originally captured one month earlier. Primarily young-of-the-year (YOY) brook trout and rainbow trout moved through the two-way weirs. The majority of movement through the weirs was in the upstream direction (74 %) and the majority of the downstream movement (66 %) occurred during the latter third of the monitoring period (July and August). The results of this study may be indicative of responses by salmonids to infertile conditions. The salmonid biomass in streams that are relatively infertile may be at carrying capacity with respect to the chemical characteristics or productivity of the water, but the physical habitat is not likely used to its full potential when compared to more productive waterbodies. Therefore, habitat use and movement patterns exhibited by salmonids inhabiting small infertile streams may be less common, but it may also be the natural response by salmonids that encounter these conditions across their geographic range.

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Chapter 1: General Introduction

Trouts, charrs, salmons, and whitefishes, collectively known as salmonids, can be found on many continents of the world where conditions permit. Salmonids comprise approximately 70 different species of fishes that dominate the northern waters of North America, Europe, and Asia and are among the most commercially important fishes (Scott and Crossman, 1973; Paxton and Eschmeyer, 1998). Consequently, many are introduced outside their native range, including Africa, South America, Australia, New Zealand, New Guinea, and India (Page and Burr, 1991). Water temperature has been the single most important factor limiting their distribution (Griffith, 1993; Scott and Crossman, 1973). For convenience, the future use of the word 'salmonid' will not include the whitefishes because of differences in life cycle or life history characteristics when compared to salmon, trout, and charr (subfamily *Salmoninae*) (Griffith, 1993).

Salmonids require cool to cold (1 - 20 °C) freshwater to fulfill part or all of their life cycles (Meehan, 1991; Griffith, 1993), with few exceptions (McDowall, 1987; Gross, 1987). Many salmonids are anadromous, meaning they mature at sea in saltwater and then migrate into streams to spawn in freshwater. Similarly, potadromous salmonids migrate into streams from freshwater lakes to spawn. Still others, complete each stage of their entire life cycles (spawning, rearing, incubation, and adult feeding) in either a freshwater lake or stream, and are referred to as lake-resident or stream-resident salmonids (Bjornn and Reiser, 1991; Raleigh, 1982).

Reproduction by salmonids occurs in the spring or fall of the year, requiring habitat features including freshwater, permeable substrate, and a constant supply of clean, cool to cold (1 – 20 °C), moderate to well-oxygenated (> 10 mg/L) water (Griffith, 1993). Substrate composition determines substrate permeability (Bjornn and Reiser, 1991). Low substrate permeability causes embryo wastes to become concentrated in

the spawning area, and reduces the ability of young fish to emerge from the spawning site. Substrate permeability is enhanced when the females dig depressions (known as redds) where they deposit the eggs. The females form depressions by flipping their tail vigorously in the area, causing the finer materials to be carried away (Moyle, 1993; Bjornn and Reiser, 1991). In streams, the constant supply of fresh water is also beneficial in reducing the concentration of embryo wastes, and supplies the embryos with oxygenated water. Redds are often formed by female salmonids at the end of pools, where the hydraulics or hydraulic gradient provides the necessary pressure to maintain a constant flow through the substrate (Bjornn and Reiser, 1991). Some species, such as brook trout (*Salvelinus fontinalis*), brown trout (*Salmo trutta*), and chum (*Oncorhynchus keta*), pink (*Oncorhynchus gorbuscha*), and sockeye salmon (*Oncorhynchus nerka*) often choose areas with groundwater seepage. Groundwater seepage helps to prevent low oxygen concentration, freezing, deposition of fine sediments in the redd, and high acidity (Bjornn and Reiser, 1991; Quinn, 1995). Choosing groundwater seepage sites for redds has also been documented for brook trout spawning in lakes, where water interchange would otherwise be quite low compared to most stream spawning sites (Curry and Noakes, 1995).

Success of incubation greatly depends on the timing of spawning in relation to climatic events during the period of incubation (Bjornn and Reiser, 1991). Water level and temperature fluctuations can greatly affect the date of emergence after incubation and the success of the incubation process. Young eggs are less efficient at respiration, and therefore, if water quality is good during the early stages of development, there are greater chances of survival. However, if suspended sediment is high, which is typical after freshets or high rainfall events that increase stream discharges, the spaces between the substrate may fill and reduce water exchange. Therefore, freshets occurring soon after egg deposition would be more harmful to developing eggs in

streams (Bjornn and Reiser, 1991). Stream water temperature is another important factor controlled by climate. The water temperature is negatively related to the length of the incubation period (Bjornn and Reiser, 1991). An early emergence may lead to high mortalities if the conditions for feeding are not suitable at the time of emergence (i.e. too cold) (Bjornn and Reiser, 1991; Griffith, 1993). Conversely, if emergence is late, the shortened season for feeding could increase winter related mortality, since reaching a certain minimum size by the fall of the year is critical for survival during winter (Cunjak and Power, 1986; Schindler, 1999). Salmonids are usually two to three centimetres long before entering the open-water environment (Thorpe, 1987).

The exodus or emigration of salmonids from the stream environment, to mature in the larger waterbody of a sea or lake, is linked to genetics and habitat conditions (Randall *et al.*, 1987; Nordeng, 1983; Thorpe, 1994; Meehan and Bjornn, 1991). Habitat conditions include: water temperature, current velocity, depths, undercut banks, shade, food availability, and structural complexity from submerged wood and large substrate (Murphy and Meehan, 1991; Meehan and Bjornn, 1991). Upon emerging from the redds, species such as chum and pink salmon migrate out of streams almost immediately regardless of habitat conditions, indicative of a strong genetic control (Meehan and Bjornn, 1991). Rainbow trout (*Oncorhynchus mykiss*) utilize the streams to a very limited degree, leaving the stream within the first couple of years (Ryther, 1997), similar to coho (*Oncorhynchus kisutch*), chinook (*Oncorhynchus tshawytscha*), masu (*Oncorhynchus masou*), and Atlantic salmon (*Salmo salar*) (Allen, 1969; Thorpe, 1987). Genetics is believed to control the size these species must attain before emigration, and the habitat conditions control the time it takes to reach a particular size (Meehan and Bjornn, 1991; Thorpe, 1987). Additionally, brown trout, sockeye salmon, brook trout, and Arctic charr (*Salvelinus alpinus*) may remain in streams for their entire lives (Ryther, 1997; Gross, 1987), and emigration is believed to occur as a result of

limited or unfavourable habitat conditions (Thorpe, 1987; Ryther, 1997).

Small headwater tributaries are used by young salmonids because they allow for increased growth (Naiman *et al.*, 1987; Erkinaro and Niemela, 1995) and lower mortality rates when compared to lake or sea habitat (Thorpe, 1994; Ryther, 1997). In streams, young salmonids spend less time and energy foraging since aquatic insects are carried downstream in the current (Griffith, 1993; Shirvell, 1994; Biro *et al.*, 1997; Fausch, 1984). Additionally, mortality is lower in small streams compared to the lake because the cool to cold temperatures, and the relatively infertile habitat, cannot support many large piscivorous or fish-eating fish (Moyle, 1993; Thorpe, 1987). The benefits of increased growth and survival are believed to be the primary reasons why salmonids usually spawn in association with tributary habitat. That is, if salmonids do not spawn in the tributaries, they usually spawn in and around the inlets (Scott and Crossman, 1973; Quinn, 1995; Thorpe, 1987). Even young-of-the-year (YOY) spawned along the lakeshore seem to be attracted to tributary habitat. In Meach Lake, Ontario, the majority (80%) of the estimated young-of-the-year (YOY) population of brook trout from the lake, made their way into small inlet tributaries within the first year (Curry *et al.*, 1997).

As stream size increases, predictable changes occur in the physical, chemical, and biological features of the habitat, which are related to the local geology and climate (Griffith, 1993; Marcus *et al.*, 1990; Gordon *et al.*, 1992; Meehan, 1991). The physical, chemical, and biological changes are a result of differences in shading, water volume and temperature, and dissolved solids or specific conductivity of the water. The sections of stream at the highest elevation, known as the headwaters, are typically cold, low in dissolved solids, well shaded, and small (low in volume of water) (Griffith, 1993; Murphy and Meehan, 1991; Gordon *et al.*, 1992). Shade helps to maintain the cool to cold stream water temperatures in headwater streams. In headwaters, the organic material that supports the lower trophic-level organisms comes from two main sources.

One source is from the groundwater. The precipitation percolates into the ground and leaches minerals from the layer of soil over the bedrock, known as the overburden, and the soil composition of the overburden contributes to the chemical characteristics of the water (Griffith, 1993; Murphy and Meehan, 1991; Gordon *et al.*, 1992). The specific conductivity (uS/cm at 25 °C) or dissolved solids in the water can be used as a coarse index of biological productivity (Scarnecchia and Bergersen, 1987; Kwak and Waters, 1997). The second source of organic material comes from the surrounding canopy or riparian vegetation, referred to as allochthonous materials (leaves, wood, and terrestrial insects). Leaves and wood from the riparian vegetation fall into the streams, decompose, and are collected by aquatic insects for food (Vannote *et al.*, 1980; Griffith, 1993). Terrestrial insects that fall into the stream from the riparian vegetation, along with the aquatic insects produced within the stream, help to support top-predators like salmonids. Moving downstream from the headwaters, a third source of organic material becomes increasingly important, which is referred to as autochthonous material. Autochthonous material is the production of plant material within the stream by photosynthesis. Autochthonous food sources are insignificant in headwaters due to the high level of shading, but the influence from the riparian vegetation becomes progressively reduced as stream size increases (Griffith, 1993; Murphy and Meehan, 1991). The coalescing of smaller streams into larger streams results in increases in dissolved solids, stream volume or discharge, and water temperture. In turn, species diversity increases since wider ranges in temperature provide more aquatic organisms with preferred temperatures, and the biomass of various organisms (fish and aquatic insects) increase since there is more living space and increased productivity (Griffith, 1993; Murphy and Meehan, 1991).

Location within a watershed affects salmonid feeding behaviour in predictable ways since the productivity or food availability increases with increasing stream size

(Vannote *et al.*, 1980). In general, stream salmonids are opportunistic feeders (Scott and Crossman, 1973; Murphy and Meehan, 1991), feeding on aquatic insects or stream macroinvertebrates, terrestrial insects, and sometimes other fish including the young of their own species (Biro *et al.*, 1997; Moyle, 1993). Organisms are taken directly off the bottom, from the water surface, and in the water column or stream drift (referred to as drift-feeding). Stream salmonids are normally observed drift-feeding (Marcus *et al.*, 1990; Allen, 1969). In small, cold, highly shaded streams, terrestrial insects form the bulk of the diet for stream salmonids possibly due to frequent periods of low water velocities and associated low amounts of aquatic insects in the stream drift, reducing the benefit of drift-feeding (Northcote, 1992; Griffith, 1993; Marcus *et al.*, 1990). In headwater streams, salmonids are invertivores since they mainly feed on the stream and terrestrial insects or invertebrates, in medium streams there is usually a mixture of invertivores and piscivores (fish-eaters), and in large rivers there may also be planktivores feeding on autotrophic plankton production (Vannote *et al.*, 1980). When conditions in small streams become unsuitable, stream salmonids have been observed moving downstream fo feed in larger rivers in the fall and early winter (Meehan and Bjornn, 1991).

During summer feeding is the highest priority for stream salmonids, invoking territorial behaviour. Territorial behaviour, involving the defense of a particular area, provides the necessary supply of food items where short forays are made from a central location (Fausch and White, 1981; Biro *et al.*, 1997; Gowan *et al.*, 1994; Chapman, 1966). Defending a territory of adequate size increases the probability that sufficient food will be carried to a stream salmonid, while decreasing energy loss from searching (Griffith, 1993). The better territories are usually defended by the dominant fish in the hierarchy. Though dominance is mainly dependent on fish size (Grant and Kramer, 1990; Griffith, 1993), both prior residence and individual aggressiveness have been

determining factors in some studies (Mense, 1975; Grant, 1990; Jenkins, 1969). As well, the selection of feeding locations depends on the size or age of the fish since smaller fish consume smaller prey (Grant *et al.*, 1998; Cunjak and Green, 1983).

The territorial behaviour or habitat use of stream salmonids during summer is affected by changes in various aspects of the stream habitat conditions (Murphy and Meehan, 1991; Allen, 1969; Griffith, 1993; Gowan *et al.*, 1994; Grant, 1990). Changes in stream habitat features that affect the available space or food in a given area of stream will effect a stream salmonid's feeding behaviour. Stream discharge and water-column velocity are two dominant features that affect territorial behaviour by stream salmonids (Bjornn and Reiser, 1991; Kohler and Hubert, 1993; Shirvell, 1994). A given volume of water has a finite number of territories for salmonids of a particular size (Allen, 1969; Grant *et al.*, 1998), and faster stream velocities increase the likelihood of dislodging stream macroinvertebrates and increasing the amount of aquatic insects in the drift (Baker and Coon, 1997; Griffith, 1993). Stream salmonids located in more productive feeding locations require smaller territories and are less aggressive in their defense (Bjornn and Reiser, 1991; Murphy and Meehan, 1991; Marcus *et al.*, 1990). As the summer progresses, stream discharges usually decrease in conjunction with the fish becoming larger, increasing the need for greater volumes of food at a time when levels of aquatic insects in the drift are decreasing (Northcote, 1992). In turn, decreases in food availability induces an increase in territory size and aggressive behaviour by stream salmonids (Murphy and Meehan, 1991; Bjornn and Reiser, 1991; Marcus *et al.*, 1990; Grant and Kramer, 1990). If habitat conditions continue to worsen, at some point territorial behaviour will not be advantageous and may cause emigration as a density-dependent response (Northcote, 1992; Allen, 1969; Power, 1980), or induce wide-range foraging behaviour typical of salmonids inhabiting lakes (Young *et al.*, 1997; Biro *et al.*, 1997).

Stream salmonids also change their habitat use patterns when living sympatrically with other salmonids, referred to as habitat partitioning (Fausch and White, 1981,1986; Allen, 1969; Marcus *et al.*, 1990; Sale, 1979). Habitat partitioning has been documented often, and is believed to be a result of either competitive interactions or simply mutual avoidance to reduce deleterious effects from aggressive interference behaviour (Grossman *et al.*, 1998; Lacroix *et al.*, 1995; Cunjak and Green, 1983). Habitat partitioning often results in one or more species using a narrower range of habitat features when sympatric compared to allopatric. For example, brook trout living allopatrically often occupy areas in pools with a wide range of depths, velocities, and cover, compared to when sympatric with many other species including brown trout and rainbow trout (Cunjak and Green, 1983; Power, 1980). When sympatric with brown trout and rainbow trout, brook trout moved to low velocity areas in pools associated with minor cover (Waters, 1983). In a study by Cunjak and Green (1983), brook trout shifted to mainly deep pools with cover when sympatric with rainbow trout. Similarly, allopatric Atlantic salmon tend to occupy shallow to deepwater positions in streams compared to mainly deepwater positions while living sympatrically with brown trout (Allen, 1969). Although the effects of habitat partitioning depend on the species involved, usually the two, or more, salmonid species develop a mutual coexistence after adjustments in habitat use (Cunjak and Green, 1983; Griffith, 1993; Marcus *et al.*, 1990).

Stream salmonids are not territorial during the winter since metabolism is low, and as a consequence they attempt to conserve energy by moving to locations providing refuge from the current, predators, and cold temperatures (Cunjak, 1996; Bjornn and Reiser, 1991; Griffith, 1993; Northcote, 1997). Deep pools, large woody debris, coarse substrate, undercut banks, and areas with groundwater seepage are some of the more typical habitat features preferred during the lethargic state of stream salmonids throughout the winter months (Griffith, 1993; Keith *et al.*, 1998). Stream

salmonids penetrate into the interstitial spaces between rocks and boulders, or form groups in slow, deep water near cover (Griffith, 1993). The association with deep pools and structure are believed to protect stream salmonids from the current, predators, and sections of stream that may freeze to the bottom (Cunjak, 1996; Griffith, 1993).

Aside from the recognized movement by salmonids during seasonal shifts in habitat use, migrations during changes in life cycle, and reproduction, they are generally believed to be sedentary or restricted in their movements while inhabiting streams (Griffith, 1993; Riley *et al.*, 1992; Northcote, 1992; Young, 1994; Gowan *et al.*, 1994). If territorial behaviour is common, minimal movement would be a reasonable expectation. Restricted movement has been observed in many studies where re-captures of stream salmonids were made within, or near to, the same pools where they were originally captured (Riley *et al.*, 1992; Gowan and Fausch, 1996; Hildebrand and Kershner, 2000). Restricted movement has been documented for brook trout (Leclerc and Power, 1980; Saunders and Smith, 1962; Shetter, 1968), masu salmon (Nakano *et al.*, 1990, as cited in Northcote, 1992), cutthroat trout (*Oncorhynchus clarki*) (Miller, 1957; Heggenes *et al.*, 1991; Hildebrand and Kershner, 2000), rainbow trout (Klein, 1974, as cited in Young, 1996; Cargill, 1980; Matthews *et al.*, 1994), and brown trout (Bachman, 1984; Harcup *et al.*, 1984; Clapp *et al.*, 1990; Solomon and Templeton, 1976; Hesthagen, 1990; Heggenes, 1988; Bunnell *et al.*, 1998; Matthews *et al.*, 1994). The numerous studies that support the theory that stream salmonids are restricted in their movements has lead some to refer it as a paradigm (Gowan *et al.*, 1994), meaning that restricted movement is generally regarded as standard behaviour by stream salmonids.

Stream salmonids will move, if possible, when conditions are unfavourable (Ryther, 1997; Grant *et al.*, 1998; Northcote, 1992; Bjornn and Reiser, 1991; Funk, 1955; Thorpe, 1987; Grant, 1990; Riley *et al.*, 1992; Biro *et al.*, 1997), but the movement observed in some studies is sometimes not as easily explained (Northcote,

1992). Movement that does not seem to be related to limited or unfavourable habitat conditions may be indicative of an individual preference (Northcote, 1992; McCormick *et al.*, 1998; Gowan *et al.*, 1994; Young, 1996). In other words, mobile salmonids within streams may be colonizers or explorers irregardless of habitat conditions (Northcote, 1992; McCormick *et al.*, 1998). McLaughlin *et al.* (1994) studied the feeding behaviour of YOY brook trout, immediately after emergence from spawning sites in three tributaries to the Credit River, Ontario. Individual brook trout, almost exclusively, were observed to behave as either sedentary or mobile foragers (McLaughlin *et al.*, 1994). In recent years, movement by small to large fractions of stream salmonid populations has been regularly confirmed through the use of radio-telemetry (Hildebrand and Kershner, 2000). Consequently, more research is needed to determine the reasons behind stream salmonid movements since it is becoming evident that mobility by stream salmonids is more common than generally understood.

Knowledge of how and when fish utilize habitat is an essential component of insuring that the habitat is protected from potential impacts of land-use (Lewis et al., 1996). Small streams along the north shore of Lake Superior are known to be used by salmonids for part or all of their life cycle (Goodyear *et al.*, 1982; Newman and Dubois, 1997). However, until recently very little was known about the physical, chemical, and biological characteristics of many small streams along the north shore of Lake Superior. This study focussed on collecting information on small Lake Superior tributaries of Nipigon Bay for two main reasons. First, land-use activities (primarily forest harvesting) may have adverse impacts on small streams (Jones et al., 1996; Meehan, 1991). An improved understanding of the characteristics of the streams is required to improve planning to ensure that the integrity of the streams is protected. Second, small streams in the area are used by brook trout that spend some of their life in Lake Superior, commonly known as "coasters" (Newman and Dubois, 1997). The decline in numbers

of coasters around Lake Superior has led to an increased interest in protection and rehabilitation of these populations (Newman and Dubois, 1997). Knowledge about the type of small stream habitat that is associated with these populations and how the fish interact with other, introduced salmonids is required to facilitate efforts to protect and enhance coaster populations.

The overall objectives of this study, were to determine the characteristics of the stream habitat occupied by brook trout and rainbow trout and the length of stream generally occupied by each species. In addition, I examined the differences in habitat use and length of stream used by brook trout, in the presence and absence of rainbow trout. The specific objectives are addressed in two studies presented in chapter four and five.

Chapter four presents the results of three experiments conducted in order to assess habitat use by brook trout and rainbow trout in streams, during the summer. The work in Chapter four tests the hypothesis that brook trout are associated with some degree of cover or structure while rainbow trout select open areas. I predicted that the catch-per-unit-effort (CUE) of brook trout and rainbow trout for traps placed in specific locations within pools, and the biomass of brook trout and rainbow trout within entire pools, would be higher in relation to various habitat features in pools. Additionally, I predicted that increases in brook trout biomass and decreases in rainbow trout biomass would result if the structural complexity of pools were increased through an experimental manipulation of pools previously low in structural complexity.

Chapter five presents the results of three experiments that were conducted to assess movement or the general length of stream used by brook trout and rainbow trout in my study streams, during the summer. The work in Chapter five tests the hypothesis that brook trout and rainbow trout in streams use small areas or territories. I predicted that capture frequencies of tagged brook trout and rainbow trout would be high for areas

near previous capture locations. Additionally, I predicted that movement, if observed, would be related to age or size of fish, stream discharge, water temperature, and time of year (i.e. beginning, middle, or end of the summer).

Chapter 2: General Methods

This section describes the general procedures and methods used to determine various chemical, physical, and fish community characteristics of the streams in this study. The chemical and physical information collected included stream discharge, water temperature, water chemistry, and water surface slope. Fish information was collected by capturing, sampling, tagging, and aging fish from different locations throughout the study.

Stream discharge was measured in each stream after rainfall events or after short periods of dry weather, using a portable water flow meter (Marsh-McBirney Inc., Flo-Mate model 2000). A tape measure was placed perpendicular to the stream flow to form a transect for collecting the necessary information. Depth was recorded at five to twenty increments along each transect, depending on the transect width; less points were used for narrow transects. Velocity values were taken at 60 % of the depth using the flow meter probe attached to the top-adjusting wading rod (Marsh-McBirney Inc.). These velocity and depth values were then used to calculate the discharges, in cubic metres per second (Appendix A, Formula A-1) (Bain and Stevenson, 1999). The same transect locations were used on a consistent basis for each stream.

Stream temperature data was collected in each stream using temperature probes (Onset Computer, model XT 1 Stowaway Tidbit). They were placed in areas with constant flow, and out of direct sunlight. Temperatures were recorded between June 17th and August 30th in all streams. In the Little Cypress River, temperatures were also recorded from August 30th in 1997 to June 17th in 1998. The probes were set to record every 4.8 hours starting at 1800 hours (i.e. 5 times/24 hours).

In the spring (May 20th) of 1997, oxygen, pH, and conductivity were determined for each of the streams using a dissolved oxygen meter (YSI Inc., model 57), a pH

meter (Hanna Instruments, model HI-9024), and a conductivity meter (Hanna Instruments, model HI-9033), respectively. During the spring (June 18th) and summer (August 6th) of 1998, water samples from each stream were sent to a laboratory (Lakehead University's Center for Analytical Services) and analysed for total alkalinity, total dissolved nitrogen, and total dissolved phosphorous.

Water surface slope (known as the energy gradient) was measured in each stream, using an auto level (Nikon, model AX-1) and a standard survey rod. Starting from where the streams discharge into Lake Superior (known as the mouth) to approximately 1 km upstream, the stream-center distance and concurrent changes in the elevation of the water surface, were measured. The total elevation change was divided by the total distance to determine the change in elevation of the water surface per kilometre of stream (Bain and Stevenson, 1999).

Electrofishing was used to sample fish in these streams. Each pool or pool and riffle combined (pool-riffle unit) was electrofished separately. A two-person team sampled each area once, moving in an upstream direction and using a generator powered backpack electrofisher (Smith-Root, Inc., model 15B). One person electrofished while the other person carried a bucket, tied to their waist. Both individuals were equipped with a net for capturing fish and a block-seine net was placed upstream of each sampling location to contain the fish.

Minnow traps were used to determine various fish community relationships in these streams. Standard 'Gee' minnow traps (50 cm long by 23 cm wide, with 0.64 cm mesh) were used. The minnow trap entrances were enlarged to 4 cm in diameter, in order to accommodate entry of the largest possible fish; no greater than 25 cm was expected. Ten to fifteen traps were set at one time for a given location. A 60 mL measurment of catfood, placed within a perforated plastic bag, was used as bait for

each trap. Traps were placed between 1000 and 1400 hours and processed approximately 24 hours later, for a maximum of three consecutive sets. Trapping occurred during July and August of 1997.

Sampling of fish included the recording of capture location, date, time, species, fork length, weight, PIT tag number (tagging described below), and fin clips (fin removal). Each fish was returned to the capture location.

Fish tagging was performed using Passive Integrated Transformer (PIT) tags (Biomark Inc., model AVID2023) as unique identifiers of salmonids. They were 2 x 12 mm glass-embedded tags. The tags were inserted into the body cavity under the left pectoral fin, using a 12-guage needle (Biomark Inc.) designed for this procedure. Brook trout and rainbow trout with fork lengths of 7 cm, or greater, were tagged. A tagging mortality experiment was run during the first four months of this study, at the Dorion Fish Hatchery, to determine expected mortality rates for this new tagging procedure (see Appendix C, Dataset C-1).

Fish scales and finrays were collected from each salmonid caught during electrofishing between September 23rd and October 7th, 1997. Otoliths were obtained from fish that were killed by electrofishing throughout the study. Scales were pressed on to acetate slides and a subset of finrays was embedded in epoxy, sliced, and mounted on glass slides. Otoliths were sent away for preparation. Aging assessments were done for scales, finrays, and otoliths by experienced personnel working for the Ontario Ministry of Natural Resources.

Chapter 3: Study Area

The three streams of this study are located within close proximity to each other, along the north shore of Lake Superior. They are, from west to east, Dublin Creek, MacInnes Creek, and the Little Cypress River. The straight-line distance between Dublin Creek and the Little Cypress River is approximately 10 km. They are located approximately 8 km east and 10 km south (48°56', 87°55') of the northern-most tip of Lake Superior (Figure 3.01).

Each stream is within the Lake Superior drainage basin and tributary to Nipigon Bay, Lake Superior. The drainage areas, from west to east, are approximately 24, 12, and 12 km^2, respectively. Elevations of around 550 m above mean sea level are common in the area of this study, which includes the Kama Hills bordering the western-most drainage (Dublin Creek). The average water surface slopes (known as the energy gradient) were 34.7, 17.9 and 17.7 m/km for Dublin Creek, MacInnes Creek, and the Little Cypress River, respectively. The watersheds are located within the Superior Highlands Ecoregion (Wickware and Rubec, 1989). The area is dominated by bedrock mainly of felsic-igneous rocks and the overburden is typically < 1 m thick, with many bare rock knobs (Gartner, 1979). Dublin Creek and the Little Cypress River have 5 and 8 m high waterfalls, respectively; located approximately 1 km upstream from where each stream discharges into Lake Superior.

This region typically receives around 600 mm of rain and 2300 mm of snowfall, annually. Normal average air temperatures during the winter, spring, summer, and fall are -14.1, 1.2, 15.7, and 4.4 °C, respectively (Environment Canada data, 1961-90, at Cameron Falls (49°09', 88°21')).

Structural complexity within these streams is provided by unembedded substrate, overhanging and submerged woody debris, undercut banks, and overhanging

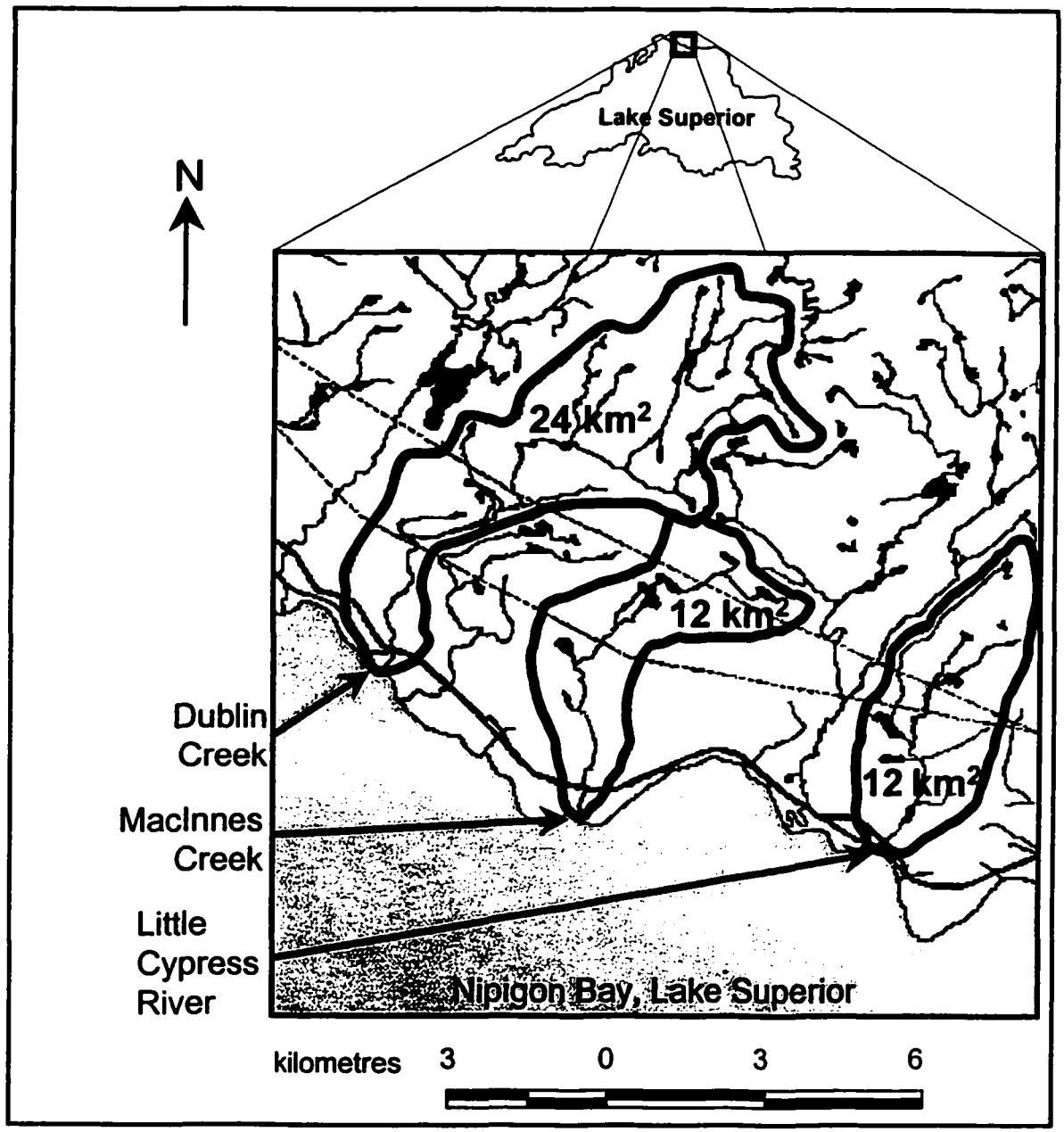

Figure 3.01: The study streams and associated watersheds on the north shore of Lake Superior.

vegetation. The streams are also characterized by alternating pools and riffles, and infrequent beaver activity. Rooted aquatic vegetation is scarce, but filamentous algae is abundant in localized areas during extended periods of summer base-flows.

In each of the study streams, specific conductivity, total alkalinity, total phosphorous, and total nitrogen were low (Table 3.01). In addition, pH values were all slightly basic, and oxygen levels were moderately high.

Each stream supports naturally reproducing brook trout (*Salvelinus fontinalis*) and rainbow trout (*Oncorhynchus mykiss*), with very few other species. MacInnes Creek and the Little Cypress River also support naturally reproducing coho salmon (*Oncorhynchus kisutch*). No coho salmon juveniles were caught in Dublin Creek but adults were seen in the system during the fall of 1997. The brook trout are indigenous to this area (MacCrimmon and Campbell, 1969), rainbow trout were introduced around 1895 (Scott and Crossman, 1973), and coho salmon were introduced in the late 1960's (Fausch and White, 1981). Populations of brook trout live above the barriers of Dublin Creek and the Little Cypress River. Other species encountered infrequently were brook stickleback (*Culaea inconstans*) in the Little Cypress River, *Phoxinus* spp. (northern redbelly or finescale dace) and sculpin (*Cottidae spp.*) in MacInnes Creek, a white sucker (*Catostomus commersoni*) above barriers in Dublin Creek, and long-nose dace (*Rhinichthys cataractae*) below barriers in the Little Cypress River. No additional species have been identified in these streams by others sampling the fish communities in recent years (Stephenson, 1998).

3.1 Study Sites

Waterfalls located on Dublin Creek and the Little Cypress River block the movement of salmonids to the upper sections of these streams. No waterfall or

Table 3.01: Chemical characteristics of the study streams.

Characteristic	Date	Dublin	MacInnes	Little Cypress
pH	Spring 1997	7.3	7.6	7.2
Dissolved Oxygen (mg/L)	Spring 1997	11.5	11.5	11.0
Specific Conductivity (*u*S/cm)	Spring 1997	24.0	70.0	49.0
Average Temperature (°C)	Summer 1997	15.3	12.9	13.8
	Summer 1998	17.7	16.3	16.8
Total Nitrogen (mg/L)	Spring 1998	0.270	0.240	0.336
	Summer 1998	0.395	0.166	0.268
Total Phosphorous (mg/L)	Spring 1998	< det. limit	< det. limit	< det. limit
	Summer 1998	< det. limit	0.006	0.007
Total Alkalinity (mg/L)	Spring 1998	42.1	107.3	68.8
	Summer 1998	61.7	164.0	93.5

migratory barrier is located on MacInnes Creek. The barriers provided me with five separate stream sections, in setting the design for the study. The five stream sections will be referred to as Dublin Upper, Dublin Lower, MacInnes, Little Cypress Upper, and Little Cypress Lower. In this manuscript, the word Upper, by itself, will collectively refer to stream sections above barriers. The word Lower, by itself, will collectively refer to stream sections below barriers.

Chapter 4: Habitat Use

4.1 Introduction

During the summer, stream salmonids associate with various physical habitat features in order to increase feeding efficiency while, at the same time, avoiding predation (Thorpe, 1994; Baker and Coon, 1997). Habitat features associated with salmonids include unembedded substrate, logs, undercut banks, deep pools, turbulent water, and shading from overhanging debris or vegetation (Murphy and Meehan, 1991; Meehan and Bjornn, 1991). In general, stream salmonids use velocity shelters close to faster water where foraging takes place (Allen, 1969; Fausch and White, 1981; Baker and Coon, 1997). Each species also seems to favour some specific habitat features. For example, brook trout use velocity shelters in pools providing some degree of structure or instream cover (Raleigh, 1982; Cunjak and Green, 1983; Power, 1980). In contrast, rainbow trout use open areas in faster current velocities, typical of riffle areas (Cunjak and Green, 1983; Keith *et al.*, 1998). Essentially, habitat use is a trade-off between predation avoidance and growth (Bjornn and Reiser, 1991), and each species has adapted their own methods in response.

Physical, chemical, and biological characteristics of streams within a species' geographic range result in many variations in the factors that influence habitat use by stream salmonids (Clarkson and Wilson, 1995; Gordon *et al.*, 1992; Meehan, 1991; Marcus *et al.*, 1990; Griffith, 1993). A stream salmonid is capable of adapting in several ways to the changing conditions that they experience across their geographic range. For example, Wilzbach *et al.* (1986) found that cutthroat trout in high elevation streams abandon their normal preference for instream cover due to the associated low productivity of high elevation streams (Scarnecchia and Bergersen, 1987; Young, 1996). Feeding becomes a priority over predator avoidance for young salmonids in high

elevation streams since reaching a minimum size by the winter is critical to survival during the typically harsh winters (Schindler, 1999). On a different note, if preferred physical habitat is limited, stream salmonids may use less desirable habitat (Baker and Coon, 1997). Flebbe and Dolloff (1995) found that brook trout in streams with high amounts of large woody debris (LWD) were distributed roughly in proportion to the availability of LWD, but large boulders serve as an effective surrogate in the absence of LWD. Furthermore, the low levels of LWD in another stream made it difficult to detect a trend in physical habitat preferences by brook trout (Flebbe and Dolloff, 1995). Therefore, habitat use by a particular salmonid may vary from one stream to another as they adapt to changes in habitat conditions (Griffith, 1993; Thorpe, 1994; Stoneman and Jones, 2000; Baker and Coon, 1997).

Stream salmonids may also alter habitat use patterns when living with other salmonids (sympatric), referred to as habitat partitioning (Fausch and White, 1981,1986; Allen, 1969; Marcus *et al.*, 1990; Sale, 1979). Habitat partitioning has been demonstrated in several studies, and is believed to be a result of either competitive interactions or adaptive segregation to reduce deleterious effects from aggressive interference behaviour (Grossman *et al.*, 1998; Lacroix *et al.*, 1995; Cunjak and Green, 1983). In either case, sympatric conditions usually result in one or all salmonid species becoming less diversified in their use of various features of the habitat (Chapman, 1978). For example, brook trout living by themselves (allopatric) often occupy areas in pools with a wide range of depths, velocities, and instream cover, compared to sympatric conditions (Cunjak and Green, 1983; Power, 1980). Under sympatric conditions with rainbow trout, brook trout shifted to mainly deep pools with instream cover (Cunjak and Green, 1983). Habitat partitioning usually leads to mutual coexistence after some minor adjustments in habitat use by the species sharing sections of stream (Cunjak and Green, 1983; Griffith, 1993; Marcus *et al.*, 1990).

In this study, I designed various experiments to determine if a relationship existed between brook trout and rainbow trout, and physical features of their stream habitat during the summer. I also examined the relationship between brook trout and habitat features, in the presence and absence of rainbow trout.

In the first part of the study, I examined the association between brook trout and rainbow trout abundance, measured as catch rates and biomass, and the physical features of the pool habitat in which they were captured. I tested the hypothesis that brook trout abundance would be higher in pools that provide some degree of instream cover or structure while rainbow trout abundance would be higher in open areas. Therefore, I predicted higher catch rates of brook trout in areas of pools associated with structure or instream cover, and higher catch rates of rainbow trout in open areas of pools. I also predicted higher biomass of brook trout and lower biomass of rainbow trout in pools with high amounts of structure or instream cover. Under sympatric conditions I predicted that brook trout would be associated with a narrower range in amounts of structure or instream cover compared to allopatric conditions.

The second part of the study involved testing responses of brook trout and rainbow trout to habitat modifications. I was testing the hypothesis that brook trout abundance would be higher in pools with instream cover and rainbow trout abundance would be higher in open areas. Therefore, I predicted that increases in structure or instream cover in pools would result in an increase in brook trout biomass within modified pools, while concurrently decreasing rainbow trout biomass.

4.2 Methods

Location

In this section of the study, the methods in Part I were employed in each of the five stream sections (see Study Sites). In Part II, only MacInnes and Dublin Lower were involved.

Part I

In July and August of 1997, minnow traps were placed over-night in specific locations within pools (see General Methods). Specific locations were selected based on the structural complexity of stream habitat within a 1 m radius of the location. The amount of structure or instream cover surrounding each trap location was evaluated subjectively, based on the presence of a combination of unembedded rocks, water depth, submerged and overhanging woody debris, undercut banks, and overhanging vegetation. Traps were set in locations categorized as no, low, and high structure. Different locations within pools were sampled simultaneously to reduce the possibility of drawing fish from one location in a pool to another, in response to the bait. Fish were sampled and tagged (see General Methods).

The catch-per-unit-effort (CUE) for each trap was calculated by summing the weight for each species and dividing by the set duration, yielding grams-per-hour (g/hr) values for each one-night set. Values of CUE were pooled for both Upper and Lower sections (see Study Sites). Since the values of CUE were not normally distributed, I used non-parametric statistical analyses to test the hypothesis that the distribution of CUE values for each structure class (none, low, and high) were the same, for brook trout and rainbow trout. Specifically, the hypothesis that biomass differed among groups (no, low, and high structure) was evaluated using Kruskal-Wallis tests, and between-group differences were evaluated using Mann-Whitney U tests.

During the summer of 1998, I measured salmonid biomass (g/m^3) and various habitat features for entire pools. The measurements were done for 62 Upper pools (50 in Dublin Creek, and 12 in the Little Cypress River) and 53 Lower pools (5 in Dublin Creek, 42 in MacInnes Creek, and 6 in the Little Cypress River). Salmonid biomass for a given pool was determined using one-pass electrofishing (see General Methods) during low-flow conditions (< 0.010 m^3/sec). It took four days to electrofish the Upper and Lower pools. The Upper pools were electrofished between July 6th and the 15th, and the Lower pools were electrofished between July 13th and the 21st. Block-seine nets were set at the top of pools to prevent fish from escaping each pool. The fork lengths of fish that evaded the electrofisher were estimated conservatively and added to the actual catch for each pool. To form more accurate values for each pool's brook trout or rainbow trout biomass, the fork length estimates of the fish that escaped capture within a pool were used to estimate the approximate weight for each fish. A regression model generated from the 1997 data, predicting weight from length, was used for this purpose. Weight estimates were used for no more than two fish for any pool, and were required for only 30 fish in total, for the entire 115 pools (Appendix C, Dataset C-1 and C-2).

The habitat features that were measured for each pool are listed below, along with a description of how they were measured:

1. Average Depth: Depths were measured, to the nearest centimetre, along transects perpendicular to stream flow in each pool. Transects began 1 m upstream from the lower end of the pool, and every 2 m thereafter. Sampling points were located at one-quarter, one-half, and three-quarters of the distance along each transect. The sum of all depth values, divided by the number of sampling points, yielded the average depth for each pool.

2. Pool Area: Average pool width (average of all transect widths) multiplied by the pool length, yielded pool area.
3. Pool Volume: Average pool depth multiplied by the pool length and the average pool width, yielded pool volume.
4. Percentage of Unembedded Substrate: Substrate particles located at each point along the transect with a median axis length greater than 10 cm, and less than 50 % embedded by fine sediment (silt or sand), qualified as unembedded. Ten centimetres was chosen as the minimum size for substrate that might provide enough structure to modify flow and large enough to be stable during normal stream discharges, thus, providing refuge for fish. The number of points with unembedded substrate, divided by the total number of sample points and multiplied by 100, yielded the percentage of unembedded substrate for each pool.
5. Percentage of Submerged Wood Material: The length and average diameter of logs (wood material estimated to have an average diameter greater than 5 cm) were measured. Also, the length, width, and depth of submerged stick piles, of 10 or more twigs or branches with average diameters (roughly estimated in the field) of less than 5 cm, were measured. Volumes of submerged wood were calculated using a separate formula for wood (Appendix A, Formula A-2). Volumes of submerged wood and stick piles were added together. The total volume of submerged wood, divided by the pool volume and multiplied by 100, yielded the percentage of submerged wood for each pool.
6. Average Canopy Closure: From the center of each pool, values of overstory closure measured in percent, were determined using a spherical densiometer (Forest Densiometers, model C). Measurements were taken while facing upstream, downstream, right, and left. The sum of the four measurements, divided by four, yielded the average canopy closure for each pool.

7. Percentage of Over-head cover: The total area of all habitat features providing shade within 20 cm of the pool's water surface, were measured and divided by the area of each pool to determine the over-head cover for each pool. Habitat features measured were:
    - overhanging wood: Defined as wood material not submerged. The area (length multiplied by average width) of wood above each pool surface was measured.
    - undercut banks: The length and depth of the undercut bank were measured and multiplied to give area.
    - overhanging vegetation: The length and width of the overhanging vegetation were measured and multiplied to give area.

To determine the habitat variable or set of variables that accounted for most of the variation among pools, I used principal component analyses to summarize the set of five habitat variables measured. The five variables included were: average depth, average crown closure, percentage of over-head cover, percentage of submerged wood, and percentage of unembedded substrate. I examined the correlation matrix to ensure that no variables in the analysis were strongly collinear ($r > 0.9$).

To determine if a relationship existed between pool salmonid biomass and the five habitat features, I regressed each pool's principal component score against biomass for brook trout, rainbow trout, and both species combined (when sympatric), within the pool. Each set of pools, Upper and Lower, were analysed separately. Additionally, brook trout biomass values within pools above barriers were transformed (log_{10}) to normalize the distribution and meet the assumption of linearity and homogeneity of variance for the linear regression analysis (Downing and Clark, 1997).

Part II

During July & August of 1998, a habitat manipulation experiment was conducted in Dublin Upper and MacInnes (see Study Sites). In each of these two sections, ten pools with low amounts of structural complexity were chosen. From the 10 pools, five pools were randomly selected for the habitat additions and five were left undisturbed.

Habitat additions were constructed by nailing together logs cut from living trees. The structures were weighted down by rocks with median axes lengths of between 15 and 30 cm. Vegetation (cedar boughs) was attached to the top of the structures to provide some over-head cover. Each structure included approximately 0.13 m^3 of wood, 6 to 7 rocks, and overhanging vegetation covering approximately 0.6 m^2 (Appendix B, Diagram B-1). All of the structures were installed on the same day. They were placed in the deepest section near the middle and upstream end of each pool. Structures were left for 1 month (after electrofishing), commencing in the middle of July (July 17th for Dublin Upper, and July 16th for MacInnes) and ending (electrofished) in the latter part of August (August 27th and 21st, respectively).

Before and after habitat additions, salmonid biomass was determined by one-pass electrofishing (see General Methods). Weight estimates were also used for fish that escaped capture, described in Part I. Weight estimates were used for no more than two fish for any pool, and were required for only 13 fish in total, including each set of 10 pools (Appendix C, Dataset C-3).

To determine if significant changes in brook trout or rainbow trout biomass within pools occurred, I performed a repeated-measures analysis of variance (ANOVA). The data from MacInnes Creek was analysed separately for each species. Time (before additions and one month after additions) and manipulation status (no additions and additions) were used as the main effects in the ANOVA model.

4.3 Results

Part I

In general, CUE was low using baited minnow traps (Figure 4.01(1,2,3)). Regardless of species or location above or below barriers, all median CUE values were zero, even with large sample sizes of between 135 and 141 trap-nights in areas of high habitat structure. However, for brook trout there was more variability in CUE from areas in pools with structure than without structure. The higher variability resulted in significant differences in CUE for brook trout above and below barriers ($\chi^2 = 8.43$, $p = 0.015$, and $\chi^2 = 24.14$, $p < 0.001$, respectively), as opposed to rainbow trout for which CUE did not differ significantly among the three structure classes ($\chi^2 = 5.04$, $p = 0.080$).

The 75th percentile for brook trout CUE values approached 1 g/hr for low and high habitat structure, except for low habitat structure under sympatric conditions of 0 g/hr (Figure 4.01(1,2,3)). Brook trout CUE under allopatric conditions differed significantly between no and low and no and high structure, but not between low and high structure (Table 4.01). For brook trout under sympatric conditions, high structure CUE was significantly greater than CUE for low structure, and low structure CUE was significantly different than no structure (Table 4.01).

During low flows in 1998, Upper area pool volumes were usually less than 20 m^3 (Figure 4.02(1)), and brook trout biomass within pools was often below 20 g/m^3 (Figure 4.03(1)). Pool volumes ranged between 1.9 and 46.0 m^3 and brook trout biomass within pools ranged between 0.89 to 110.00 g/m^3.

Among pools in the Upper area, a principal component analysis explained over half of the variability in the habitat features among pools using two components (Table 4.02). The first principal component, which explained 43.9 % of the variability among pools, represented a gradient of pools with high amounts of wood and over-head cover

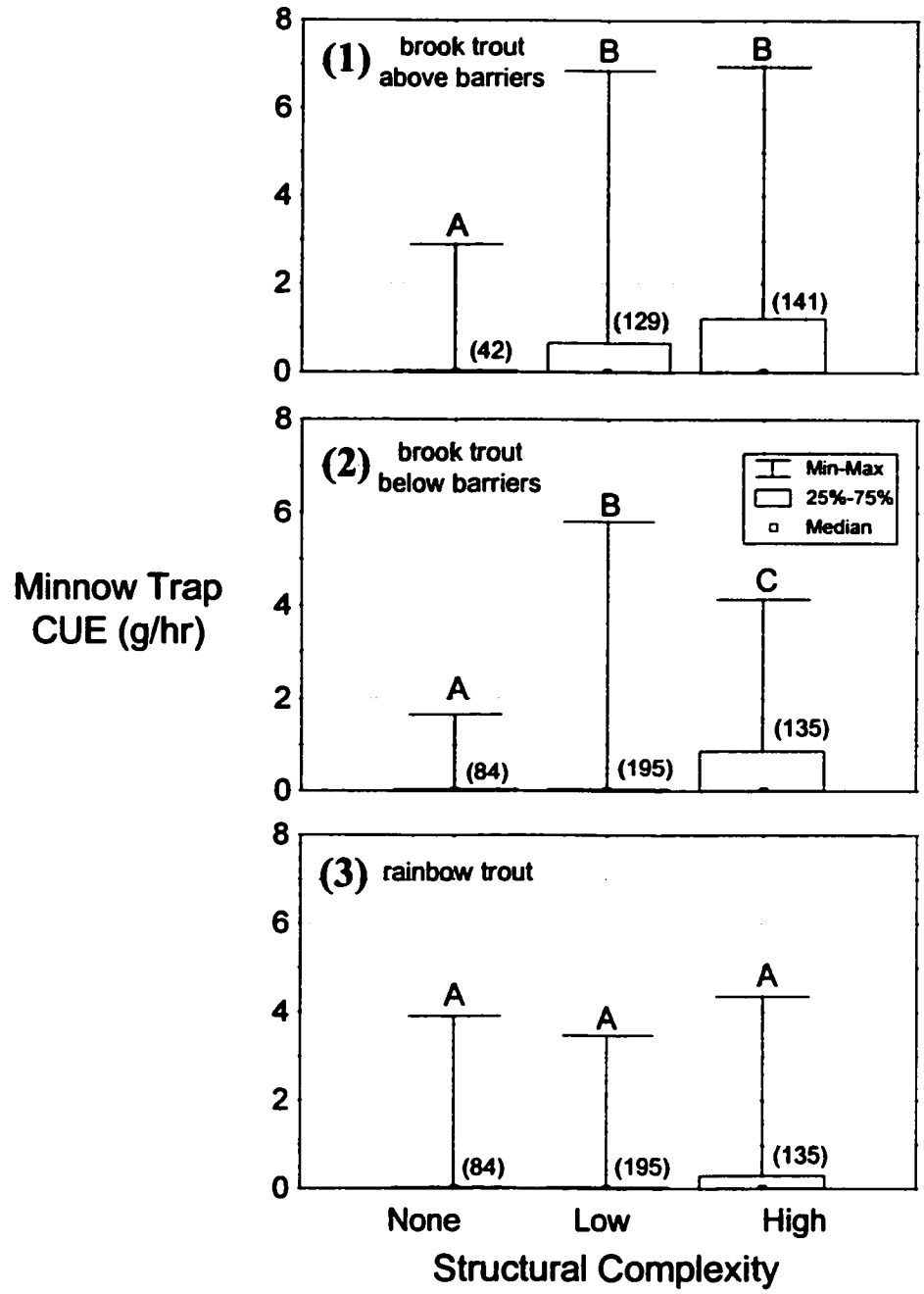

Figure 4.01: Catch-per-unit-effort (CUE) of brook trout (1 & 2) and rainbow trout (3) caught by minnow trapping in areas of no, low, and high structural complexity. Different letters denote significantly different groups. Group sample sizes are in parentheses.

Table 4.01: Mann-Whitney U analysis between CUE values (g/hr) for each combination of no (none), low, and high structure class using minnow-traps, in 1997. This analysis was done for brook trout above barriers (n=42, 129, and 141, respectively) and below (n=84, 195, and 135, respectively).

Location	Species	Comparisons	Z	p
Above	Brook Trout	None versus Low	-2.50	**0.012**
		None versus High	-2.98	**0.003**
		Low versus High	-1.21	0.224
Below	Brook Trout	None versus Low	-2.72	**0.006**
		None versus High	-4.65	**<0.001**
		Low versus High	-3.01	**0.003**
Below	Rainbow Trout	Not applicable (Kruskal-Wallis >0.05)		

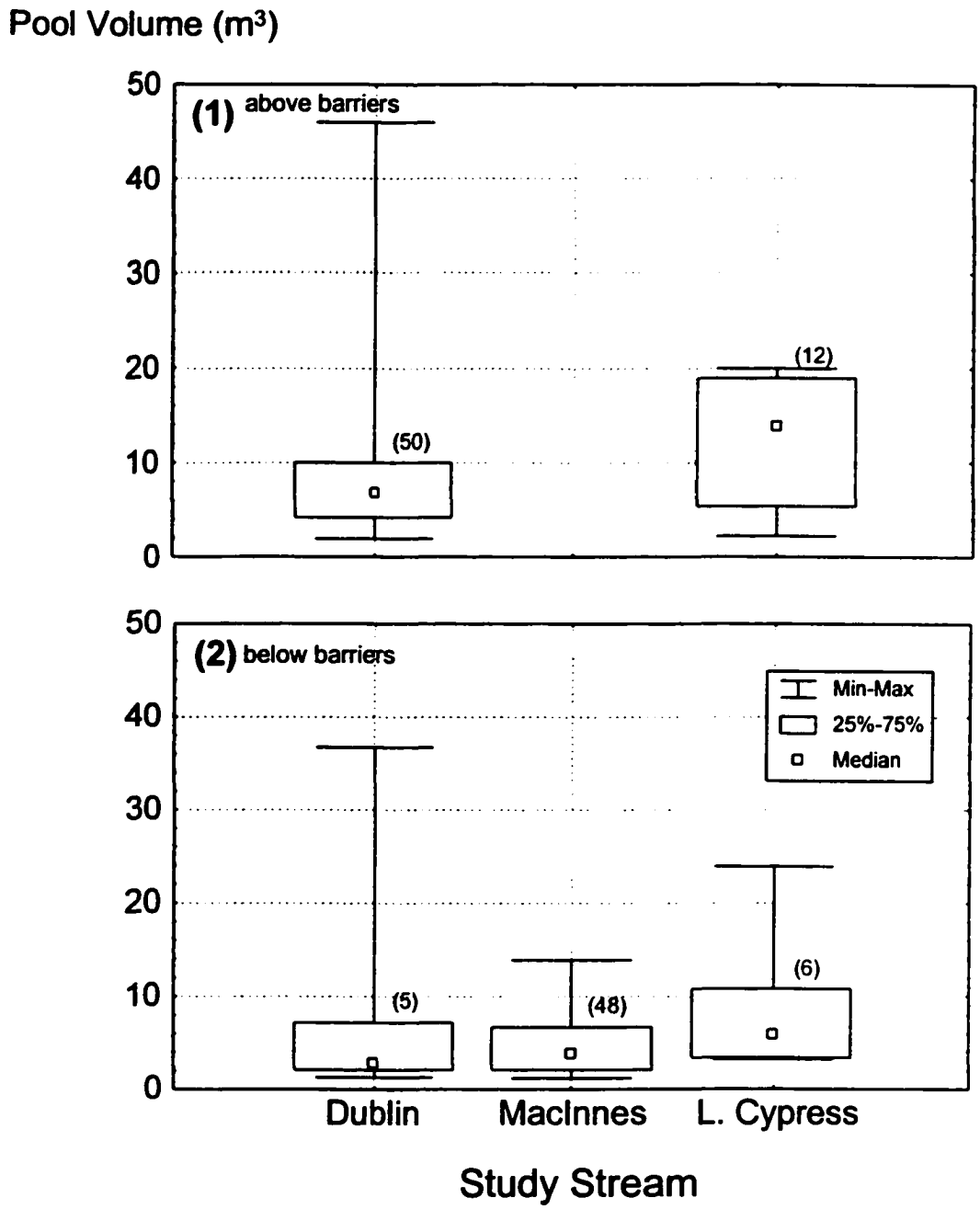

Figure 4.02: Pool volumes for each of the streams above (1) and below (2) barriers, in 1998. Group sample sizes are in parentheses.

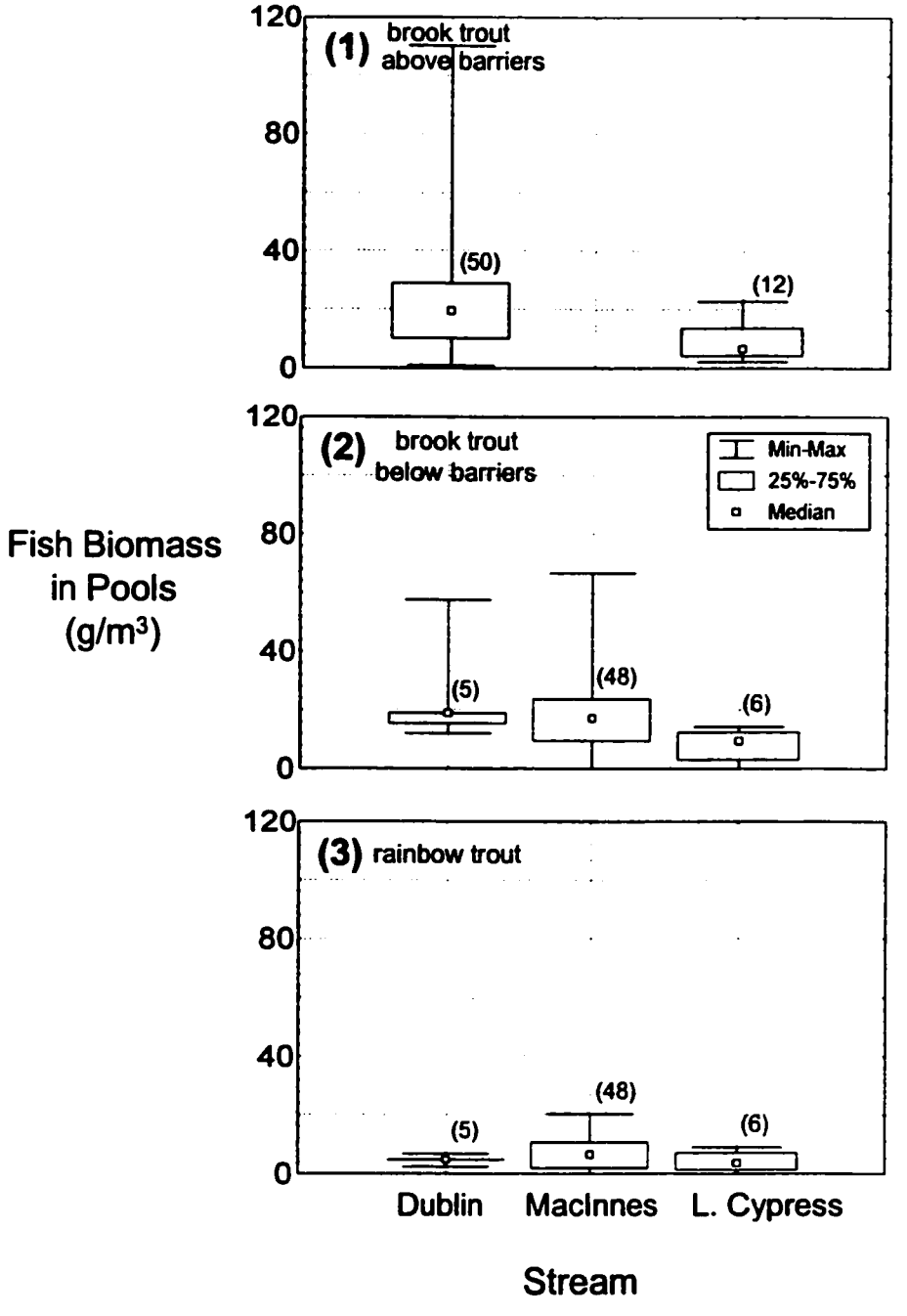

Figure 4.03: Fish biomass within pools of each stream for brook trout above (1) and below (2) barriers, and for rainbow trout (3), in 1998. Group sample sizes are in parentheses.

Table 4.02: Principal component analysis of five habitat variables from pools of Dublin Upper (n=50) and Little Cypress Upper (n=12), in 1998.

Feature	Principal Component 1	Principal Component 2
Wood Loading	**-0.725235**	-0.271885
Over-head Cover Loading	**-0.899155**	0.067704
Unembedded Rocks Loading	0.312329	0.580692
Canopy Closure Loading	**0.781178**	-0.026105
Mean Depth Loading	-0.388986	**0.764238**
Eigenvalue	2.193544	1.000449
% Variance Explained	43.8709	20.0090

to pools with high amounts of canopy closure. The second component axis, which explained 20.0 % of the variability among pools, represented a gradient of pools from shallow to deep.

Above barriers, a significant positive relationship was identified between high levels of canopy closure and brook trout biomass within pools, along with a significant negative relationship with high levels of wood and over-head cover (Figure 4.04(1)). The first principal component scores for each pool were regressed against the log of brook trout biomass within pools, yielding a significant relationship ($F = 4.577$, $p = 0.037$). The slope and intercept were significantly different from zero, and the model accounted for less than 6 % of the total variance in brook trout biomass within pools (Table 4.03).

During low flows in 1998, Lower area pools were usually less than 10 m^3 (Figure 4.02(2)), and brook trout or rainbow trout biomass within pools was often below 20 g/m^3 (Figure 4.03(2,3)). Pool volumes ranged between 1.2 and 36.7 m^3. Brook trout biomass within pools ranged between 0.00 and 66.62 g/m^3, and rainbow trout biomass within pools ranged between 0.00 and 20.39 g/m^3.

Among pools in the Lower area, a principal component analysis explained over half of the variability in habitat features among pools using two components (Table 4.04). The first principal component, which explained 31.8 % of the variability among pools, represented a gradient of pools from low to high amounts of canopy closure. The second principal component, which explained 21.1 % of the variability among pools, represented a gradient of pools from no wood to high amounts of wood.

Below barriers, a weak positive relationship was identified between high levels of canopy closure and brook trout biomass within pools (Figure 4.04(2)), but none with rainbow trout or total salmonid (brook trout and rainbow trout) biomass within pools (Table 4.03). The slope and intercept were significantly different from zero, and the

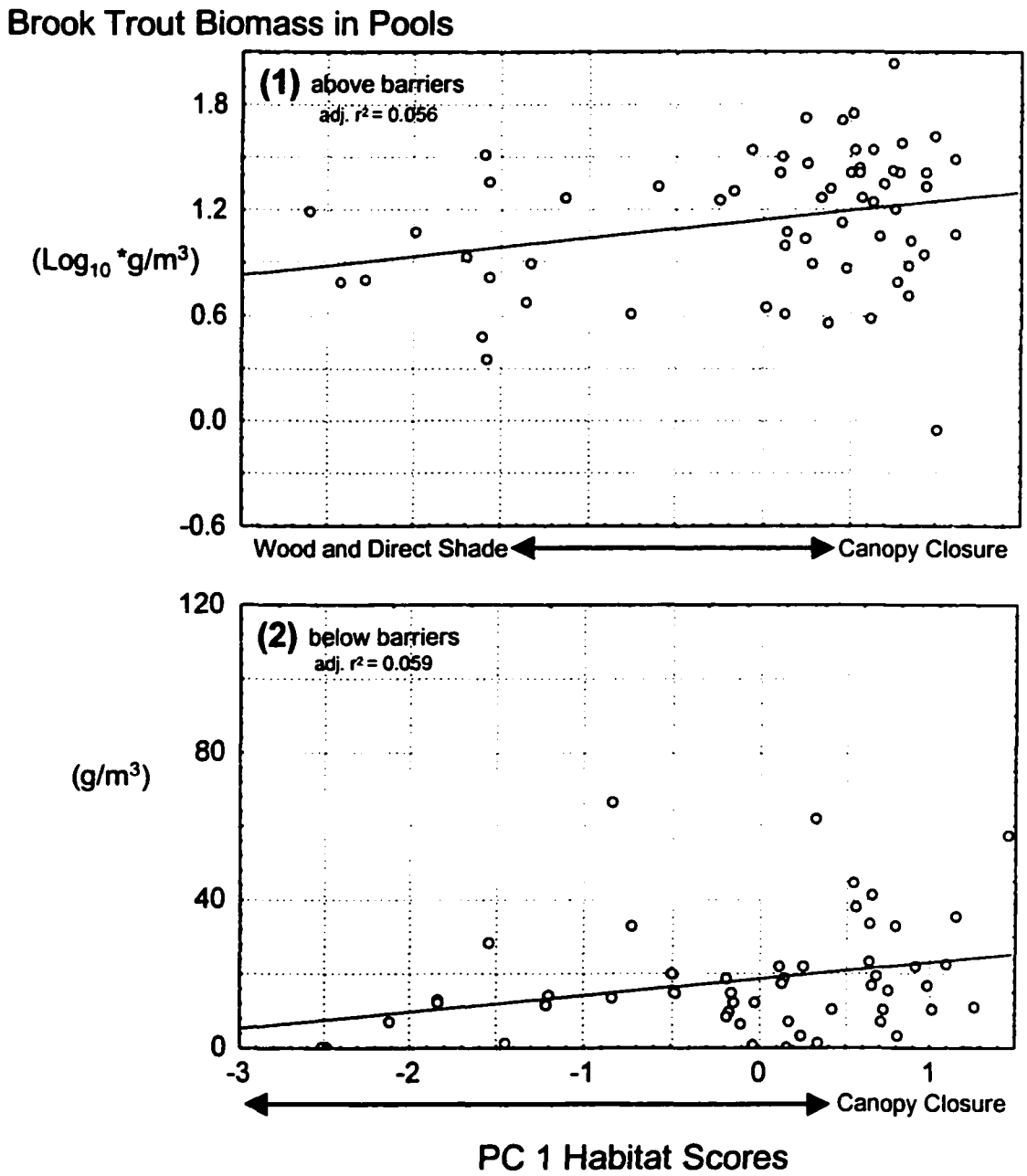

Fig 4.04: The linear relationship between Principal Component 1 (PC 1) scores for pool habitat features and brook trout biomass (g/m^3) for respective pools above (1) and below (2) barriers, in 1998. Brook trout biomass above barriers was transformed (log_{10}) (where, log_{10} * biomass = 1.1451 + 0.1051 * PC1), but not below barriers (where, biomass = 18.497 + 4.2297 * PC 1).

Table 4.03: Linear regression analysis of Principal Component 1 (PC 1) against salmonid biomass within pools, above barriers (n=62) and below (n=53), in 1998.

Location	Independent Variable	Dependent Variable	F	Sig. of F	t	Sig. of t	Adj. r^2
Above	PC 1	Brook Trout Biomass (g/m^3)	4.577	**0.037**	23.319	**<0.001**	**0.056**
Below	PC 1	All Salmonid Biomass (g/m^3)	1.385	0.245	12.287	**<0.001**	0.007
Below	PC 1	Rainbow Trout Biomass (g/m^3)	1.652	0.204	8.831	**<0.001**	0.012
Below	PC 1	Brook Trout Biomass (g/m^3)	4.253	**0.044**	9.105	**<0.001**	**0.059**

Table 4.04: Principal component analysis of five habitat variables from pools of Dublin Lower (n=5), MacInnes (n=48), and Little Cypress Lower (n=6), in 1998.

Feature	Principal Component 1	Principal Component 2
Wood Loading	-0.366495	**-0.728528**
Over-head Cover Loading	-0.571238	0.161135
Unembedded Rocks Loading	0.503007	0.525945
Canopy Closure Loading	**0.721684**	-0.324317
Mean Depth Loading	-0.595805	0.344835
Eigenvalue	1.589458	1.057428
% Variance Explained	31.7892	21.1486

model accounted for less than 6 % of the total variance in brook trout biomass within pools (Table 4.03).

Part II

One month after habitat manipulations in pools above barriers, the variability and average values of brook trout biomass decreased within both manipulated and unmanipulated pools (Figure 4.05(1)). The average brook trout biomass within manipulated pools decreased by 24 % (60 g), and the coefficient of variation decreased by 2.9 %. The average brook trout biomass within unmanipulated pools decreased by 5 % (9 g), and the coefficient of variation decreased by 23.3 %. There were no significant main effects for manipulation status (no additions and additions) or time (before additions and one month after additions), and no interaction effects (repeated-measures analysis of variance (ANOVA)) (Table 4.05).

One month after habitat manipulations in pools below barriers, the variability and average values of brook trout biomass increased within manipulated pools and decreased within unmanipulated pools (Figure 4.05(2)). The average brook trout biomass within manipulated pools increased by 80 % (48 g), but the coefficient of variation increased by 51.5 %. The average brook trout biomass within unmanipulated pools decreased by 18 % (51 g), and the coefficient of variation decreased by 66.8 %. There were no significant main effects for manipulation status (no additions and additions) or time (before additions and one month after additions), and no interaction effects (repeated-measures ANOVA) (Table 4.06).

The variability and average values of rainbow trout biomass increased within manipulated and unmanipulated pools one month after habitat manipulations in pools (Figure 4.05(3)). The average rainbow trout biomass within manipulated pools increased by 46 % (17 g), and the coefficient of variation increased by 41.6 %. The average rainbow trout biomass within unmanipulated pools increased by 47 % (20 g),

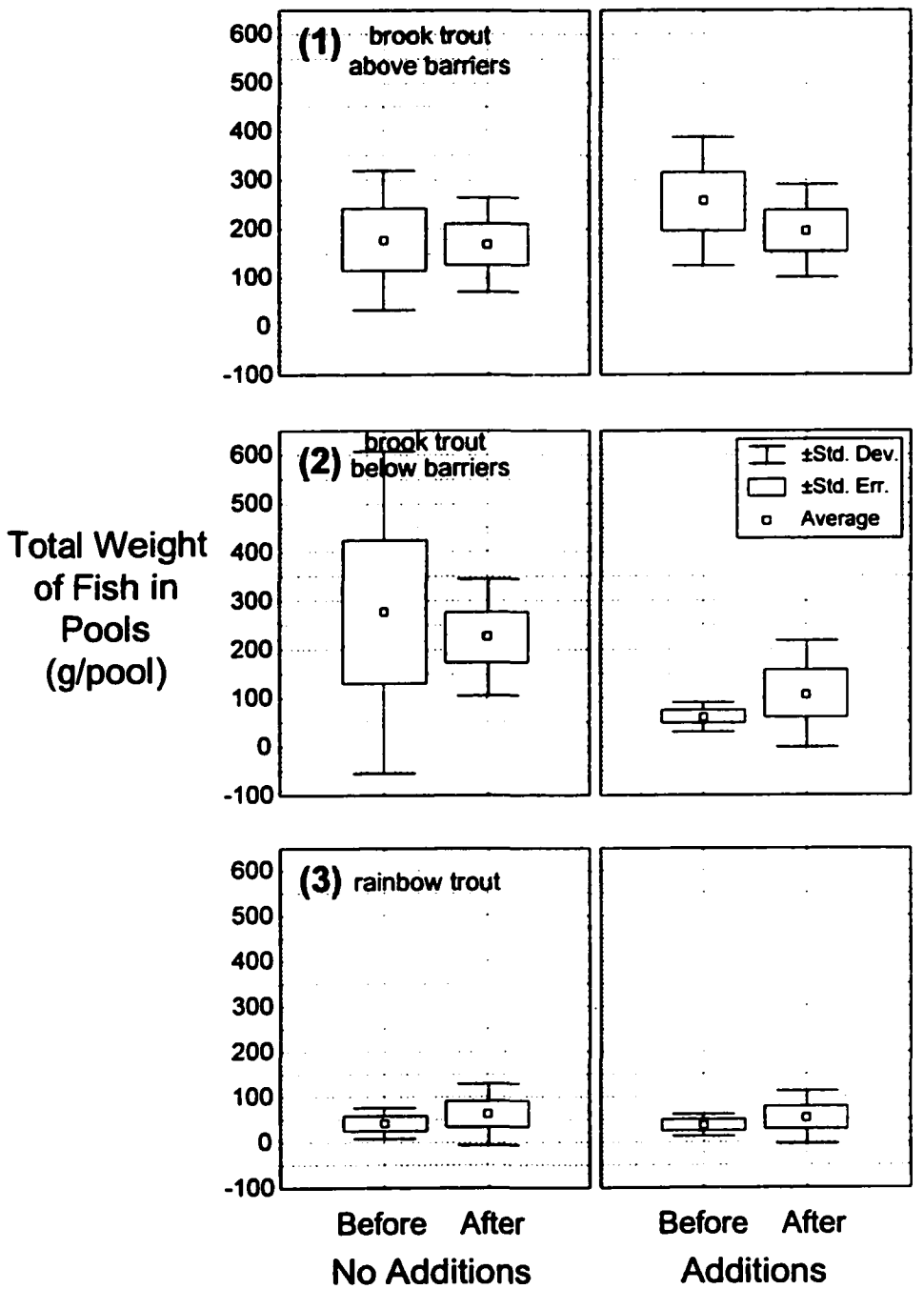

Figure 4.05: Total weight of fish for the manipulated and unmanipulated pools before and after the habitat additions for brook trout above (1) and below (2) barriers, and rainbow trout (3), in 1998.

Table 4.05: Repeated measures analysis of variance for brook trout biomass changes as part of the habitat manipulation experiment performed in Dublin Upper, in 1998.

Source of Variation	df Effect	MS Effect	df Error	MS Error	F	p-level
Manipulation (yes vs. no)	1	14526.05	8	17899.30	0.812	0.394
Time (before vs. after)	1	6020.45	8	10180.20	0.592	0.464
Manipulation by Time	1	3302.45	8	10180.20	0.324	0.585

Table 4.06: Repeated measures analysis of variance for brook trout biomass changes as part of the habitat manipulation experiment performed in MacInnes Creek, in 1998.

Source of Variation	df Effect	MS Effect	df Error	MS Error	F	p-level
Manipulation (yes vs. no)	1	138611.3	8	54170.48	2.559	0.148
Time (before vs. after)	1	11.3	8	14393.47	0.001	0.978
Manipulation by Time	1	12152.5	8	14393.47	0.844	0.385

and the coefficient of variation decreased by 28.0 %. There were no significant main effects for manipulation status (no additions and additions) or time (before additions and one month after additions), and no interaction effects (repeated-measures ANOVA) (Table 4.07).

4.4 Discussion

Results from the first part of this study indicated that there was no difference in rainbow trout CUE among specific pool locations that differed in habitat structure class. This finding was contrary to the prediction that rainbow trout would be more abundant in open areas. Studies of habitat use by rainbow trout often indicate a preference for open areas in associaton with fast current velocities, typical of riffles (Cunjak and Green, 1983; Keith *et al.*, 1998). However, riffle areas weren't sampled in this experiment because they provided very little water, and preliminary sampling in riffle areas indicated that very few fish of either species were present in these areas. When not actually in riffles, rainbow trout are usually found either at the top or bottom end of pools, where water velocities are higher than in the middle of pools (Keith *et al.*, 1998). One reason for the apparent lack of difference could be that a disproportionate number of minnow traps were placed in slower pool water, since turbulent water was lacking in these streams during the summer. If areas with turbulent or faster water are the preferred locations for rainbow trout in these streams, the low CUE for rainbow trout may be an effect of a bias in the sampling protocol towards areas low in water velocity.

Brook trout under allopatric conditions were more likely to be captured in minnow traps in areas containing at least some habitat structure or instream cover, and under sympatric conditions brook trout were captured most commonly in pools with high levels of habitat structure. The results support my prediction that brook trout would be more

Table 4.07: Repeated measures analysis of variance for rainbow trout biomass changes as part of the habitat manipulation experiment performed in MacInnes Creek, in 1998.

Source of Variation	df Effect	MS Effect	df Error	MS Error	F	p-level
Manipulation (yes vs. no)	1	186.05	8	3859.65	0.048	0.832
Time (before vs. after)	1	1711.25	8	981.85	1.743	0.223
Manipulation by Time	1	8.45	8	981.85	0.009	0.928

abundant in areas providing instream cover, and that under sympatric conditions the association with habitat structure would be more evident as a consequence of habitat partitioning with conspecifics. These findings are also consistent with observations from other studies examining brook trout habitat use in both sympatric and allopatric situations (Power, 1980; Cunjak and Green, 1983).

Providing food as bait in traps is not a common procedure used in other studies examining habitat use by stream salmonids. Baiting may cause fish to behave abnormally by drawing them into an area that they would not normally have chosen to enter otherwise. Baited minnow traps were used to reduce sampling mortality associated with electrofishing and many areas within pools were sampled simultaneously to prevent fish from being drawn to the trap from a different location in the pool. However, the number of traps that were placed in areas providing some degree of instream cover was higher than the number placed in areas with no instream cover, and may partially account for the increased variability in fish CUE observed from minnow traps placed in areas with structure. Therefore, although the results from minnow trapping were consistent with other studies examining habitat use by brook trout, further tests are needed to substantiate them.

Above and below barriers, brook trout biomass was slightly higher in pools with high amounts of canopy closure over pools. In addition, brook trout biomass was lower in pools with high amounts of wood and over-head cover in Upper areas. These findings were contrary to my predictions that brook trout would be associated with instream cover or structure, as indicated by my preliminary findings. However, high levels of canopy closure over pools might decrease predation from avian predators and help to maintain cooler temperatures. In addition, pools with high levels of canopy closure may increase food availability through increased allochthonous input in the form of leaf litter or terrestrial insects falling or landing on the water. The canopy may provide more

organic input for stream invertebrates or provide an increased input of terrestrial insects, relative to more open pools. Both factors may increase foraging opportunities for fish. An association with canopy closure over pools could be indicative of food as the limiting factor in these streams as a result of the infertile water. Therefore, canopy closure over pools may be beneficial to the brook trout in my study streams in many ways.

The weak association of brook trout with high levels of canopy closure could be partially related to the low biomass of brook trout in these streams and to differences that affect the success of avian predators. Pools with high canopy closure consisting of low shrubs like speckled or green alder, might have lower levels of predation by avian predators (Bugert *et al.*, 1991) such as kingfishers (*Megaceryle alcyon*), since kingfishers prefer an unobstructed view and perches between 20 – 40 feet above the water surface (Hamas, 1994). Kingfishers are known to be a major threat to brook trout in headwater streams (Power, 1980). In my study, I did not measure the height above the pool's surface where the canopy closed over the stream, but these differences did exist. If avian predators are a significant source of mortality, the success of predators in pools that differ in the height of canopy closure above a pool's surface may partially explain the weak associations found.

The speculation that salmonid distribution is related to food availability in the study streams is also supported by evidence that the low fertility of the water in these streams may limit the abundance of their normal aquatic food sources. Salmonid biomass in these streams ranged from 1.81 to 3.99 g/m^2, consistent with another study on Lake Superior tributaries of < 5.8 g/m^2 (Waters *et al.*, 1990). Similarly, in unproductive Idaho streams, 2 – 3 g/m^2 was reported as the upper limits for chinook salmon (Sekulich, 1980, as cited in Bjornn and Reiser, 1991), and O'Connor and Power (1976) detected 1.21 to 5.33 g/m^2 in their study of unexploited brook trout populations located in northern Quebec streams. The reported ranges in biomass, for O'Connor

and Power (1976), included four streams over a two-year span and were the lowest reported in North America at that time. Thus, the salmonid biomass in my study streams are comparable to the lowest levels reported in the literature for stream salmonids.

Low salmonid biomass in the streams studied by O'Connor and Power (1976) was believed to be primarily a result of the infertile water. O'Connor and Power (1976) found that the nutrient and ionic concentrations of the water, and the climatic conditions of their streams were very similar to northwestern Ontario. Specific conductivity (uS/cm at 25 °C) and total alkalinity (mg/L of $CaCO_3$) are chemical features of water that are often used as coarse indicators of stream productivity (Scarnecchia and Bergersen, 1987; Kwak and Waters, 1997). Correlations between water chemistry and salmonid biomass are stronger for infertile streams since the chemical properties of the water are usually the limiting factor to productivity (O'Connor and Power, 1976; Scarnecchia and Bergersen, 1987; Kwak and Waters, 1997). Specific conductivity values may reach levels greater than 600 *u*S/cm in some streams (Kwak and Waters, 1997), but < 70 uS/cm is considered low (Scarnecchia and Bergersen, 1987; O'Connor and Power, 1976). Specific conductivity values less than 70 uS/cm were seen by O'Connor and Power (1976) (<11 uS/cm) and in my study streams (<70 uS/cm). Total alkalinity values may reach levels as high as 300 mg/L, but levels below 120 mg/L are considered low (Kwak and Waters, 1997). These low total alkalinities were observed in studies by O'Connor and Power (1976) in Quebec streams (< 120 mg/L), Waters *et al.* (1990) in Lake Superior tributaries (< 83 mg/L), and in my study streams, where only MacInnes fluctuated above 120 mg/L (107 - 164 mg/L) by the middle of the summer (Table 3.01). Thus, it seems reasonable to assume that my study streams are limited by the fertility of the water to support a high abundance of aquatic life.

Results from the second part of this study indicate that there was no detectable response by either brook trout or rainbow trout to the habitat manipulations. It was predicted that brook trout biomass would increase, and rainbow trout biomass would decrease, in response to habitat structure additions under sympatric conditions. The simplest explanation for the lack of response could be that the structural features added were not preferred habitat. Although, if food is limiting in these streams instead of habitat, additions of more preferred habitat would be inconsequential. The lack of response could also be due to an insufficient length of time allocated before examining for changes. Time is one of the most important factors for monitoring responses of fish populations to habitat manipulations (Hunt, 1976). Many evaluations of habitat manipulation experiments indicate that salmonid populations may not fully respond for as many as five to seven years after treatment (Riley and Fausch, 1995). Furthermore, since a response to these additions would necessitate movement by fish, more needs to be learned about the movement patterns by brook trout and rainbow trout in these streams. For example, if movement is limited, few fish would have had the opportunity to discover the changes, and relocate if desired. Conversely, if movement is extensive and frequent, many fish likely would have encountered the manipulated habitat and made their choice to relocate, or not. Therefore, these results may be indicative of the fact that either habitat structure is not preferred or fish were not sufficiently exposed to the changes.

Based on the results presented, the distribution of brook trout in these streams during the summer could be related to the low levels of food production within these streams. Low food production is typical of small, cold, chemically dilute streams located in the temperate regions of North America (Griffith, 1993; Marcus *et al.*, 1990; Meehan, 1991). Under infertile conditions, high CUE of brook trout in baited minnow traps associated with instream cover, a preference for pools with greater inputs of

allochthonous materials, and no response to increases in stream structure that does not contribute to increased food levels in pools, would be reasonable predictions. These speculations, however, do warrant further investigations. Information regarding the diet of brook trout and rainbow trout, along with the abundance of stream macroinvertebrates or their representation in the drift, would be some of the parameters to consider. Furthermore, direct observations of fish using various features of the habitat would alleviate problems associated with some of my methods that depended on an assumption of limited physical habitat.

Knowledge regarding the temporal and spatial scale of the movements by brook trout and rainbow trout during the summer in these streams is also necessary for better interpretations of these results. Point-in-time sampling of fish that typically remain in small areas or home ranges should provide more representative information regarding habitat use, than for fish that use large home ranges and regularly move to other pools throughout the area (Riley and Fausch, 1995; Gowan *et al.*, 1994). If brook trout and rainbow trout in these streams use large areas of the stream in the summer, then they may associate with instream cover while moving from one pool to the next to avoid predation, may temporarily use areas unrepresentative of preferred habitat, and may be affected less by small additions of preferred habitat as home ranges increase in size. Therefore, although it seems logical to suggest that these results are primarily indicative of the aquatic productivity of these streams, large home ranges could also be contributing to the discrepancies. The inconsistencies that may be a result of movement patterns among the pools of these streams will be addressed in the next chapter of this thesis, Chapter five.

Chapter 5: Movement

5.1 Introduction

Aside from the recognized movement by stream salmonids during seasonal shifts in habitat use, changes in life cycle, and reproduction, they are generally believed to be sedentary or restricted in their movements while inhabiting streams (Griffith, 1993; Riley *et al.*, 1992; Northcote, 1992; Young, 1994; Gowan *et al.*, 1994). Restricted movement behaviour is based on the fact that stream salmonids are territorial, defending an area from subordinates in order to provide the necessary amount of food to meet their energetic requirements (Chapman, 1966; Gowan *et al.*, 1994; Biro *et al.*, 1997). Both laboratory (Fausch and White, 1986; Fausch, 1984) and field studies (Bachman, 1984; Morantz *et al.*, 1987; Shirvell and Dungey, 1983; Fausch and White, 1981; Jenkins, 1969; Biro, 1998) have provided convincing evidence of territorial behaviour by stream salmonids. The territorial behaviour includes, defending and feeding within specific areas where stream salmonids make short forays to capture prey items. Restricted movement behaviour has been supported by many studies reporting the relocation of stream salmonids within, or near to, the same pools where they were previously located (Riley *et al.*, 1992; Gowan and Fausch, 1996; Hildebrand and Kershner, 2000). There are studies documenting restricted movement by many species of salmonids including brook trout and rainbow trout (Klein, 1974, as cited in Young, 1996; Cargill, 1980; Matthews *et al.*, 1994). Numerous studies support the theory that stream salmonids are restricted in their movements which has lead some to refer it as a paradigm (Gowan *et al.*, 1994), meaning that restricted movement is generally regarded as standard behaviour by stream salmonids.

Stream salmonids are expected to be sedentary during the summer, but they are also expected to move if conditions within the stream environment change, becoming unfavourable (Ryther, 1997; Grant *et al.*, 1998; Northcote, 1992; Bjornn and Reiser,

1991; Thorpe, 1987; Grant, 1990; Riley *et al.*, 1992; Biro *et al.*, 1997), similar to the responses by most animals (Northcote, 1978). Movement away from unfavourable conditions in search of new opportunities is an animal's basic biological response to adversity (Taylor and Taylor, 1977). For stream salmonids, when competition exists for a territory, the larger fish within the various age or size classes are dominant, forcing smaller fish to search for other locations (Grant and Kramer, 1990; Griffith, 1993). Consequently, movement by YOY salmonids has often been observed when competition for space is believed to be a factor (Northcote, 1992; Moore and Gregory, 1988). Movement as a result of limited habitat conditions resulting in increased competition for a limited number of spaces has been observed for many species of salmonids including young rainbow trout (Bjornn, 1971; Slaney and Northcote, 1974) and brook trout (Bjornn, 1971; Riley *et al.*, 1992).

A limited food supply may also invoke wide-range foraging tactics by dominant stream salmonids (Bachman, 1984). Clapp *et al.* (1990) found that large brown trout (> 400 mm) moved between 300 and 33,000 m in the Au Sable River, Michigan, and concluded that it was likely due to a switch from drift feeding to piscivory or fish-eating. In a radio-telemetry study by Young *et al.* (1997), large (188 – 240 mm) cutthroat trout were actively foraging during the summer in the Colorado river because of what was concluded to be a patchily-distributed invertebrate supply. Gowan and Fausch (1996) concluded that the relatively slow growth rate of large brook trout emigrating from sections of streams in northern Colorado was indicative of a limited food supply. Mesick (1988) also concluded that emigration of Apache (*Oncorhynchus apache*)and brown trout was related to unfavourable feeding conditions since those emigrating were in poor condition. Thus, dominant stream salmonids may also move in response to various habitat limitations such as an inadequate food supply.

In contrast, stream salmonids may become more sedentary if pressure on the

available resources within a stream is alleviated. Flick and Webster (1975), after removing competitors of brook trout from a small New York stream located in the Adirondack Mountains, found the movement of brook trout through their weirs decreased significantly over the eight-year monitoring period. They concluded that the decreased movement was likely in response to an increase in suitable living space. Likewise, after a large pond was created before the estuary in a coastal Prince Edward Island stream, Saunders and Smith (1962) observed almost a complete cessation in the exodus of brook trout from the stream. Movement by young and old stream salmonids, depending on the circumstances, should also increase in frequency as summer progresses because fish increase in size and discharges usually decrease, resulting in a need for larger territories in a stream where less and less space is available (Chapman, 1966; Grant, 1990; Bjornn, 1971; Northcote, 1992; Allen, 1969; Power, 1980). In short, if an animal's needs are not being met, they must move or die (Thorpe, 1994).

In the last decade, movement by small to large fractions of groups or populations of stream salmonids has been regularly confirmed through the use of radio-telemetry (Hildebrand and Kershner, 2000). The regular movement observed has caused some to suggest that movement by stream salmonids may be indicative of an evolutionarily stable or innate strategy, not necessarily occurring as a result of unfavourable conditions (Northcote, 1992; McCormick *et al.*, 1998; Gowan *et al.*, 1994; Young, 1996). In other words, there may be a selective advantage for some members of a salmonid population within streams to be colonizers or explorers rather than being sedentary (Northcote, 1992; McCormick *et al.*, 1998). McLaughlin *et al.* (1994) studied the feeding behaviour of YOY brook trout, immediately after emergence from spawning sites in three tributaries to the Credit River, Ontario. Individual brook trout, almost exclusively, expressed either a sedentary or mobile feeding behaviour (McLaughlin *et al.*, 1994).

Therefore, the more accurate characterization of the movement by stream salmonids may require the use of both mobile and sedentary categories, instead of categorizing a population as either one based on the majority rule.

In this study, I designed various experiments to determine the approximate length of stream or number of consecutive pool-riffle units that brook trout and rainbow trout use, and whether any notable differences were apparent for brook trout in the presence and absence of rainbow trout. A pool-riffle unit being a pool (an area of slow moving water) and the riffle (an area of fast water, often turbulent on the surface) immediately upstream from the pool. I also designed an experiment to determine whether movement within these streams was related to changing habitat conditions and various characteristics of the brook trout and rainbow trout that were moving.

In the first part of the study, I examined the capture frequencies of previously tagged brook trout and rainbow trout in pools within the reaches of my study streams during the summer. I was testing the hypothesis that both brook trout and rainbow trout use a limited section of stream during the summer. Therefore, within short and long stream reaches I predicted high capture frequencies of previously tagged fish near their original capture pools.

The second part of the study was designed to monitor movement into and out of a Lower (see Study Sites) study-stream section. I was testing the hypothesis that the number of fish moving through the weirs over a 24 or 48-hour period would be related to some combination of size or age of fish, fish species, stream discharge, stream water temperature, and time of year (i.e. beginning, middle, or end of the summer). I predicted that movement might increase under various situations, such as increased stream discharges, higher water temperatures, and by a certain age or size class of fish.

5.2 Methods

Location

In 1997, a reach was selected within each of the five stream sections of this study (see Study Sites), and used in Part I. In 1998, a different reach was selected within Little Cypress Lower and used in Part I and Part II.

Part I

In 1997, to determine if brook trout and rainbow trout were primarily using only a few consecutive pool-riffle units during the summer, reaches averaging 100 m in length and comprising four to five pool-riffle units were selected (Table 5.01). A pool-riffle unit refers to a pool (an area of slow moving, deeper water) and the riffle (an area of fast water, often turbulent on the surface) immediately upstream from the pool. All five reaches were located upstream of the highway. Reaches within Dublin Lower, MacInnes, and Little Cypress Lower stream sections were all located approximately 200 m upstream from the highway and also approximately 200 m downstream from the first migratory barrier on both Dublin Creek and the Little Cypress River. The two Upper reaches were located between one and two kilometres upstream from a migratory barrier.

During the beginning of the summer (June 10th to July 10th) each reach was electrofished, and each fish was sampled and tagged (see General Methods). One to 2 months later (July 29th to August 7th), using the same methods, the reaches and the areas above and below each reach were electrofished. These areas were electrofished to identify the locations of brook trout and rainbow trout that had been tagged during the initial electrofishing within the reaches.

In 1998, a 466 metre-long section of stream located below barriers in the Little Cypress River was selected to determine the length of stream that was being used by the brook trout and rainbow trout in the stream. The Lower section of the Little Cypress

Table 5.01: The stream section, length, and number of pool-riffle units for each of the reaches used in 1997.

Stream Section	Reach Length	Number of Pool-Riffle Units
Dublin Upper	128.4	4
Little Cypress Upper	130.0	5
MacInnes	101.8	5
Dublin Lower	71.9	5
Little Cypress Lower	83.2	5

River is an approximately one kilometre-long section of stream between the migratory barrier and the river mouth, where the stream discharges into Lake Superior. The reach was comprised of 30 pool-riffle units. Fish movement into, and out of this section of stream was controlled by placing a two-way weir at the top and bottom of the section. Weirs were constructed using quarter-inch block-seine mesh (Appendix B: Diagram B-2). Weir 1 was located 33 m downstream from the migratory barrier, and Weir 2 was located another 466 m further downstream (Figure 5.01). The area between the weirs was electrofished at the beginning of the summer (June 24th and June 25th) and fish were sampled and tagged (see General Methods). Between the two days, a block-seine net was left overnight to keep fish in the section that was electrofished on the first day separated from the area upstream, to be electrofished on the second day. Capture of any of the tagged fish between the weirs was recorded. All fish caught in the weirs were individually sampled (see General Methods) and released to the opposite side of the weir from which they had entered. Electrofishing occurred approximately one month later (July 20th and 21st) to compare the capture locations of previously tagged brook trout and rainbow trout with their original capture pools.

Part II

With the use of the two-way weirs located in the section of the Little Cypress River (described above), I was able to monitor the movement of fish into, and out of, the area between the weirs. The weirs were inspected between June 24th and August 30th, 1998. They were checked daily, except on three occasions for elapsed periods of two days. For these three occasions, the numbers of brook trout and rainbow trout sampled on the second day were divided in half to represent each of the two sampling days. Additionally, the weirs had to be removed for a period of five days, between August 13th and the 17th, because of high water.

Salmonids caught within the weirs were given fin clips (complete fin removal) as

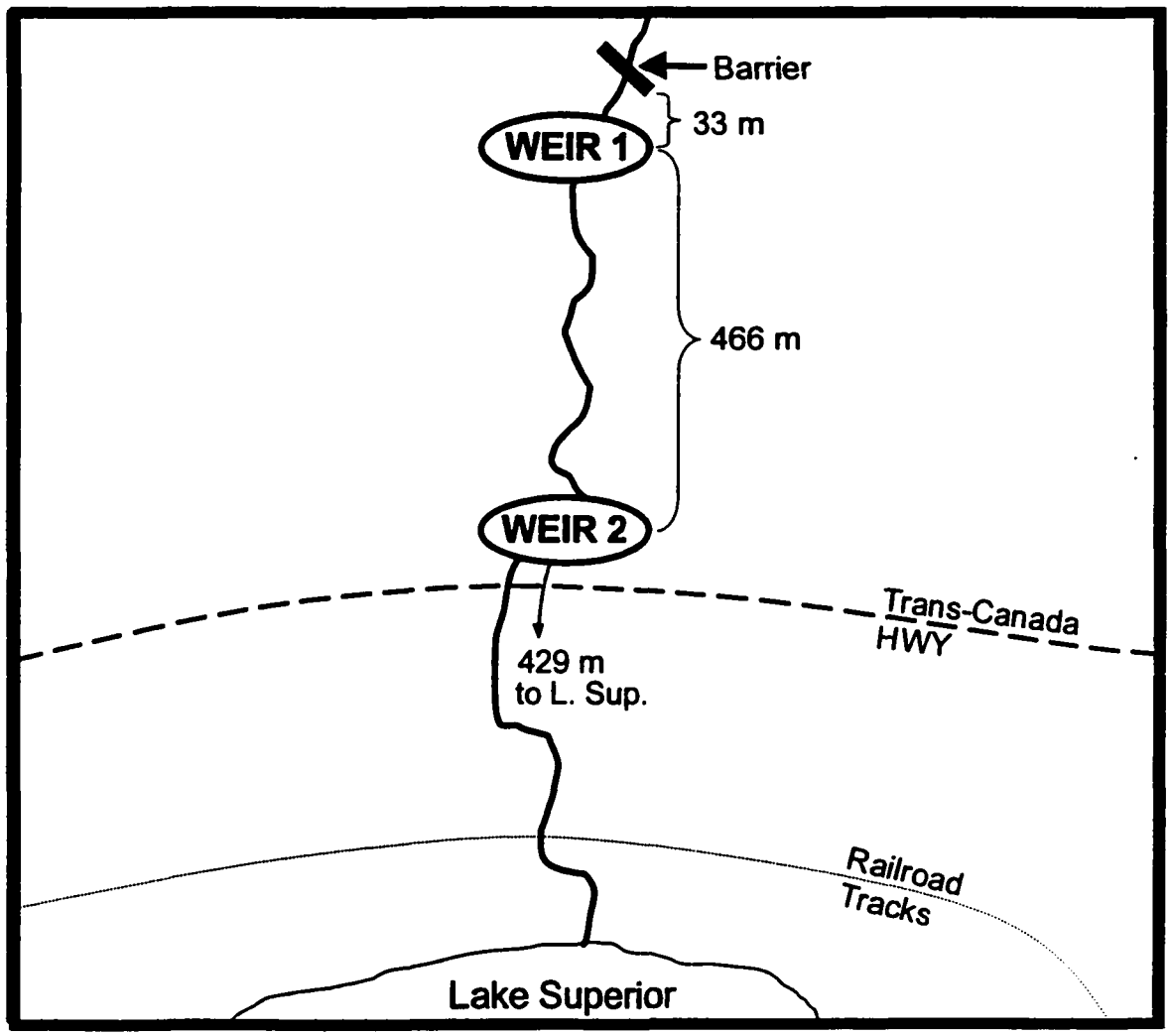

Figure 5.01: Weir locations in the Little Cypress River, during the summer of 1998.

a general identification for their most recent capture location. Those caught moving through Weir 1 were given both right and left ventral fin clips, and for Weir 2, both right and left pectoral fins were removed. Direction of movement, species, fork length, weight, date, clips, and tag identification numbers (applicable for the experiment explained in Part I, above) were recorded. In addition, a small sample of finrays and otoliths, taken throughout the summer, were kept for aging analysis (see General Methods). Fish were released to the opposite side of the weir from which they had entered.

Rainfall was measured using a standard rain gauge (All Weather Plastic Rain Gauge, model P-2000), located in a clearing next to Weir 1. Stream discharges were determined after rainfall events and after short periods of dry weather (see General Methods). To determine if a relationship existed between stream discharge and the number of brook trout and rainbow trout moving through the weirs, I used a linear regression analysis. Values of stream discharges for a particular date, were regressed against the number of brook trout and rainbow trout caught in the weirs, pooled by direction of movement over a 48-hour period. The 48-hour period encompassed the 24 hours prior to the discharges being taken, plus the next 24 hours. This period of 48 hours was used because it usually took a minimum of 24 hours for discharges to become noticeably reduced. When discharges were taken on consecutive days, only the discharge value for the second day was used for the regression analysis, to ensure that catch values were independent. The number of rainbow trout moving in the downstream direction were square-root transformed to meet the assumption of linearity and homogeneity of variance for the linear regression analysis (Downing and Clark, 1997).

Stream water temperatures were measured, in the Weir-2 pool, for the duration of the monitoring period (see General Methods). To determine if a relationship existed

between changes in the average daily stream water temperature and the number of brook trout moving through the weirs, I used a linear regression analysis. The change in average daily water temperature was regressed against the number of brook trout caught in the weirs during the 24-hour period, and pooled by direction of movement. Transformations (log (x+1)) were necessary for brook trout catches, to meet the assumption of linearity and homogeneity of variance for the linear regression analysis (Downing and Clark, 1997). Rainbow trout were not used for this analysis since the distributions were not normal, and transformations were unable to correct the problem.

In order to determine the age-classes of brook trout and rainbow trout moving through the weirs I used age-at-length data from the fall of 1997, since only a small number of aging structures (finrays and otoliths) were collected in 1998 from brook trout and rainbow trout caught in the weirs. In the fall of 1997, a large number of aging structures (finrays and scales) were collected between September 23rd and October 7th (see General Methods). This age-at-length data from 1997 was used to estimate the age-classes of fish that used the weirs in 1998. However, it is difficult to determine ages of fish caught during the summer using age-at-length data from the fall. Therefore, growth data was used to make a projection of the fork length each brook trout and rainbow trout, captured in the weirs, would likely have attained by the fall of the year. The estimation of weight is based on an assumed linear or constant growth rate. This enabled a direct comparison of age-at-length distributions from the fall of 1997 with fork length estimates projected to the fall of 1998.

To determine an appropriate growth rate for the length-projection formula (Appendix A, Formula A-3), I compiled the growth information from fish that were previously tagged (see General Methods) and later captured, in 1997. The growth information was pooled for all streams, by species. The average growth rate for brook trout and rainbow trout was 0.17 and 0.18 mm/day, respectively (Figure 5.02). To

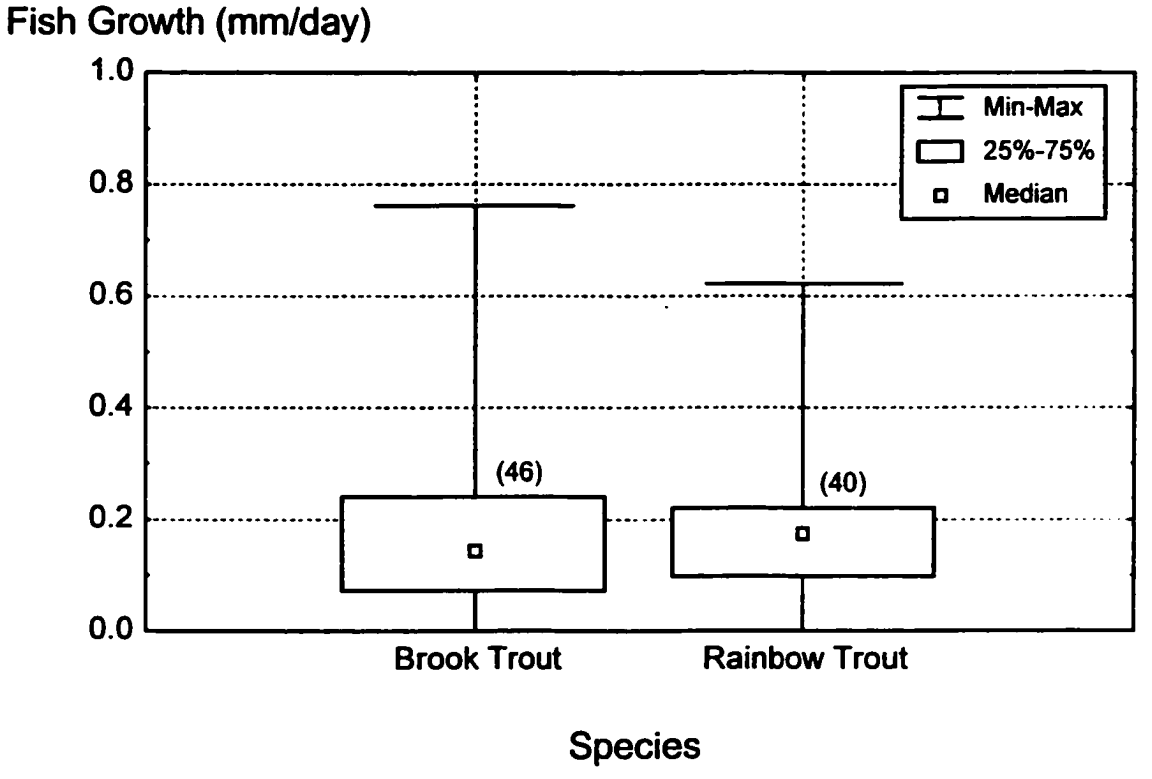

Figure 5.02: Growth-rates for brook trout and rainbow trout based on captures of PIT-tagged fish during the two years of the study. Group sample sizes are in parentheses.

ensure that the fork length projections were not underestimates, I used an above average growth rate of 0.20 mm/day in the formula, for both species (Appendix A, Formula A-3). A forward-calculation was based on the assumed daily growth rate and the difference in days, between the first-time capture date and September 30. Length projections for each fish captured in the weirs were calculated for September 30th, 1998.

5.3 Results

Part I

In 1997, there were 19 captures of 102 previously tagged brook trout and rainbow trout within the five stream reaches (Table 5.02). Only one previously tagged fish was captured outside the limits in each of three reaches. Above barriers, the average capture frequency of previously tagged brook trout was 26.1 %, compared to rates of 13.3 % for brook trout and 9.3 % for rainbow trout below barriers. A previously tagged brook trout was captured 63.5 m downstream from the Dublin Upper reach, another brook trout 300 m downstream from the Little Cypress Lower reach, and a rainbow trout 15 m downstream from the MacInnes reach.

In 1998 almost all of the captures of previously tagged brook trout (4/5) and rainbow trout (10/11) were within two pools of their initial capture pool, and none was caught leaving the area between the weirs during that period. From the total number of tagged fish, 25 % (5/20) of the brook trout and 31 % (11/36) of the rainbow trout were re-captured. Only one of each species was found farther than two pools away from their original capture pool; a brook trout was captured eight pools upstream, and a rainbow trout was captured 5 pools downstream.

Part II

Seventy-four percent of the fish captured, both brook trout and rainbow trout, were moving in an upstream direction. Of the 26 % that were caught moving

Table 5.02: Percentages of previously captured brook trout and rainbow trout, caught after repeat electrofishing within each of the reaches used in 1997, selected within each of the five stream sections of this study.

Stream Section	Species	First Sample Date	Return Date	Number Tagged	Number Recaptured	Percentage Recaptured
Dublin Upper	brook trout	Jul 10	Aug 7	22	6	27.3
Little Cypress Upper	brook trout	Jun 17	Aug 6	24	6	25
MacInnes	brook trout	Jun 12	Jul 29	6	1	16.7
Dublin Lower	brook trout	Jun 20	Jul 31	13	3	23.1
Little Cypress Lower	brook trout	Jun 10	Jul 30	2	0	0
MacInnes	rainbow trout	Jun 12	Jul 29	14	0	0
Dublin Lower	rainbow trout	Jun 20	Jul 31	13	2	15.4
Little Cypress Lower	rainbow trout	Jun 10	Jul 30	8	1	12.5

downstream three-quarters were captured in Weir 1 and primarily during the latter part of August (Figure 5.03(1,2)). A total of 341 individual brook trout and 79 individual rainbow trout were captured in the weirs (Figure 5.04), along with 39 individual coho salmon (*Oncorhynchus kisutch*), 116 individual long-nose dace (*Rhinichthys cataractae*), and 17 individual brook stickleback (*Culaea inconstans*). Twenty-five percent (44/185 – 10 (killed)) of the brook trout that moved upstream through Weir 2 were later caught at Weir 1. Approximately half (21/44) were again caught moving back downstream from above Weir 1. Another 106 unmarked brook trout moved upstream through Weir 1, having been present in the stream between the weirs at the time of weir placement. Only one rainbow trout, out of 15 that were caught moving upstream through Weir 2, was eventually caught moving upstream through Weir 1 and was not caught again.

Increased movement through the weirs was noted during increased discharges at the beginning, middle, and end of the monitoring period, but no trend was apparent with changes in the average daily water temperature (Figure 5.05). Rainbow trout catch rate was low in the weirs, particularly in the middle of the summer. The stream discharge of 0.551 m^3/sec, as measured on August 13th, required the removal of the weirs for five days. Average daily water temperatures, throughout the monitoring period, fluctuated between 13 and 19 °C.

The variability in the numbers of brook trout and rainbow trout moving through the weirs during increased discharges was greatest for brook trout moving upstream, compared to brook trout moving downstream and rainbow trout moving upstream (Figure 5.06(1,2,3)). Stream discharge measurements regressed against the numbers of brook trout moving upstream during the 48-hour period (includes the 24-hour period before and after the discharge measurements) showed a significant relationship ($F = 6.549$, $p = 0.028$). Similarly, a significant relationship was found for brook trout moving

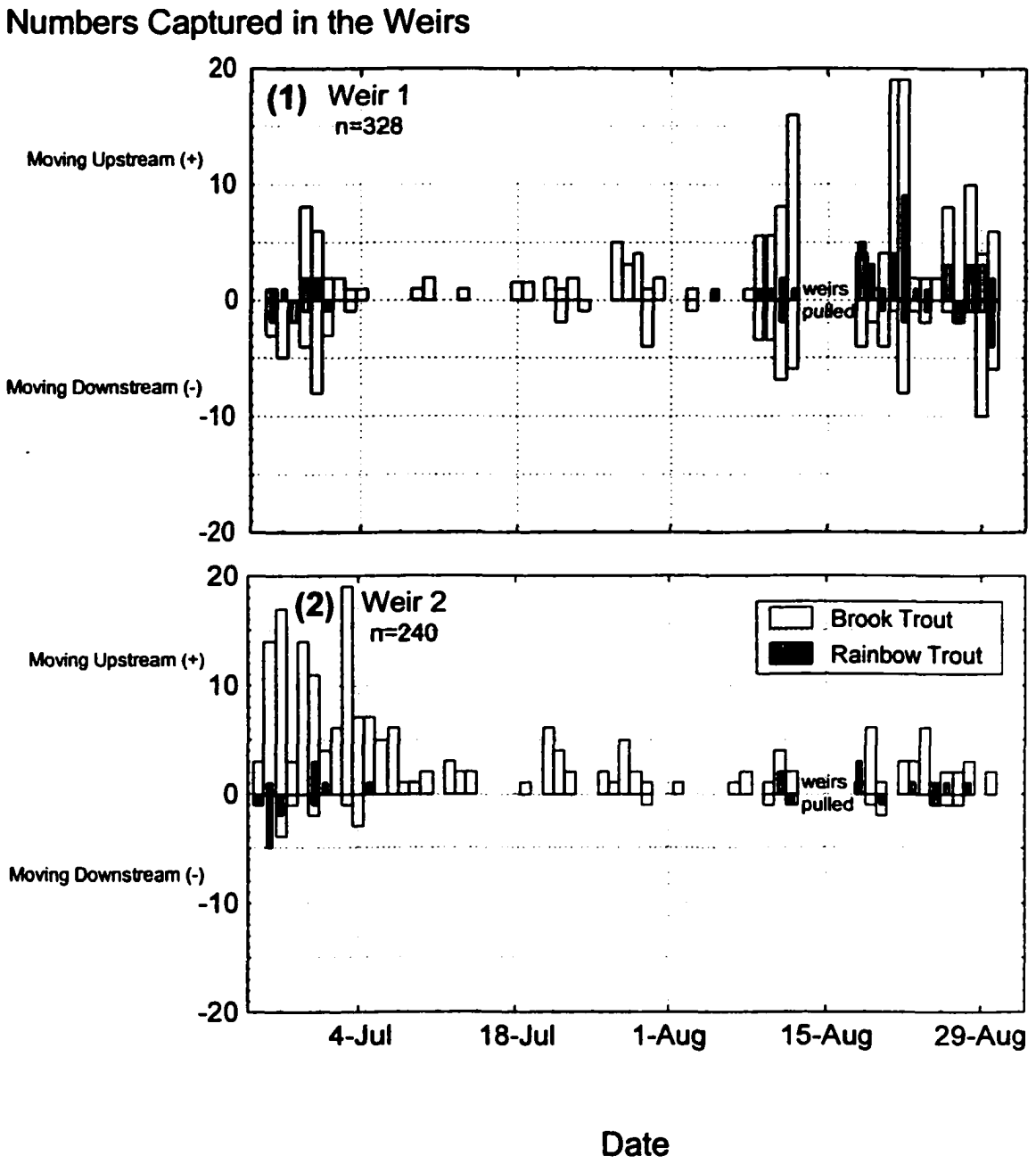

Fig 5.03: Daily movements of brook trout and rainbow trout through Weir 1 (1) and Weir 2 (2), located in the Little Cypress River in 1998. Negative values indicate downstream movement and positive values indicate upstream movement through the weirs.

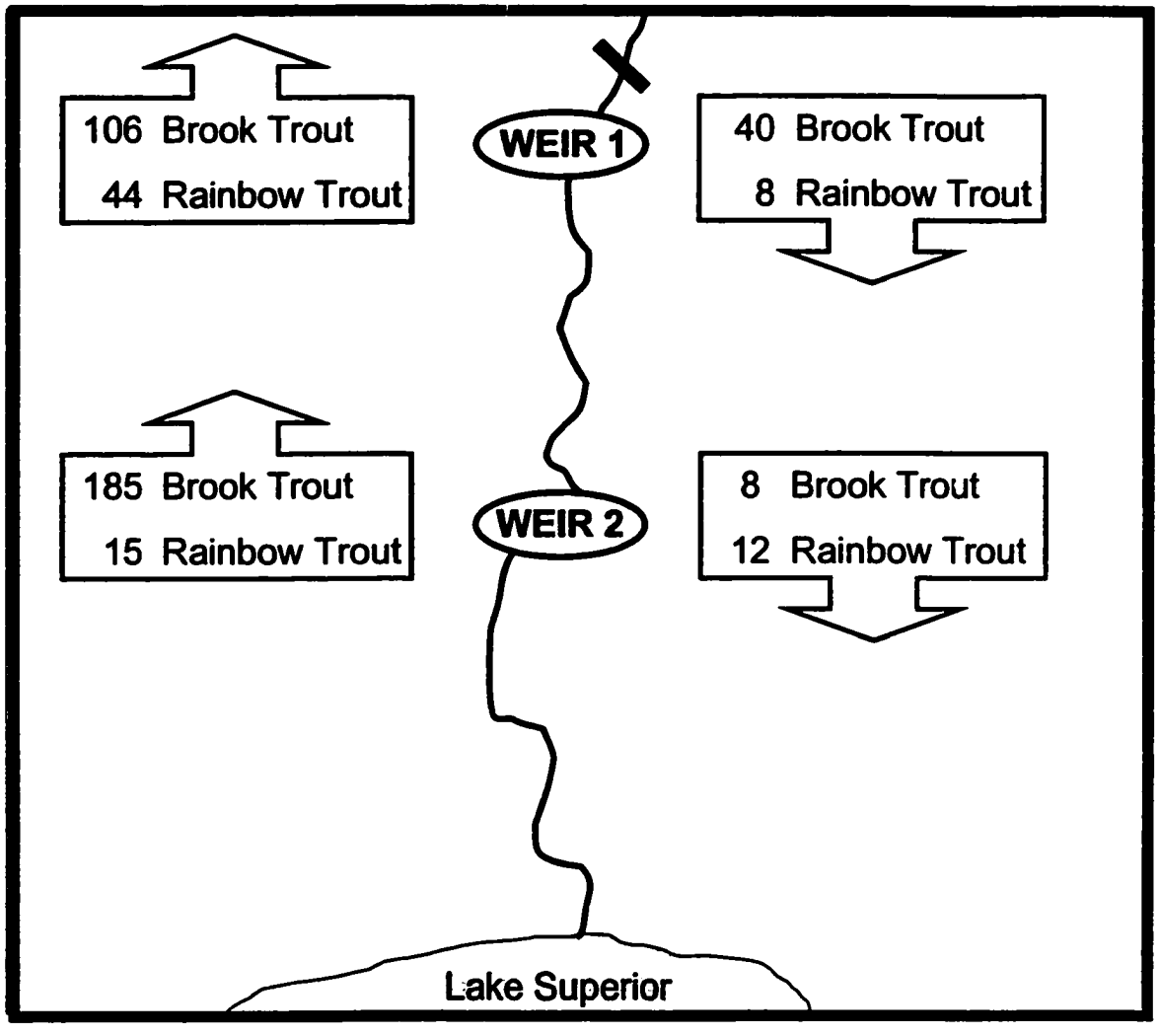

Figure 5.04: The weir locations in the Little Cypress River and respective numbers of individual (first-time captures) brook trout and rainbow trout that were caught in the weirs, for both directions.

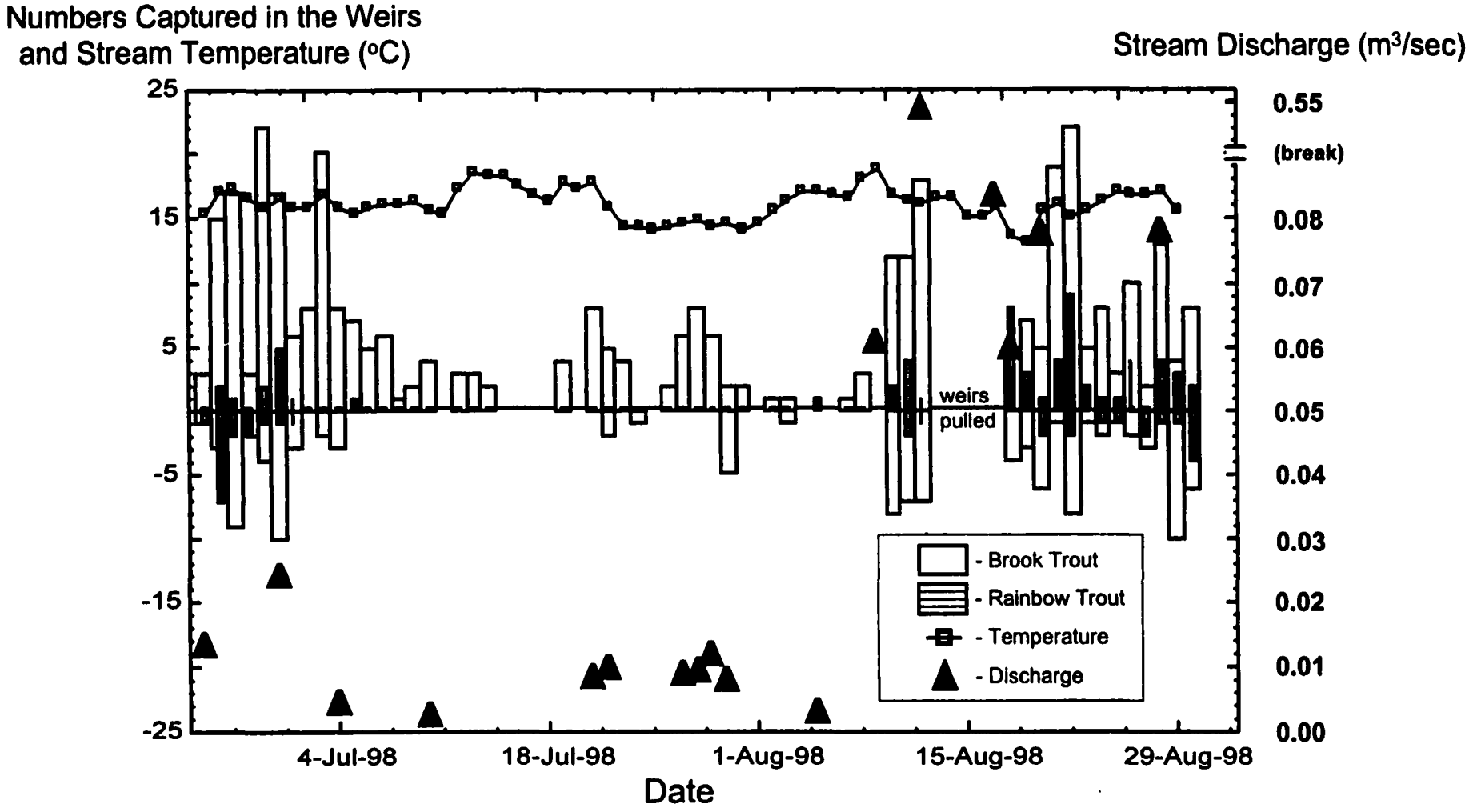

Figure 5.05: Daily weir captures of brook trout and rainbow trout, pooled by direction for the two weirs located in the Little Cypress River in 1998. Included are mean daily water temperatures and stream discharges taken during the sampling period. Positive and negative values on the y-axis indicate upstream and downstream movement, respectively. The y-axis is scaled for both water temperature and the numbers captured in the weirs.

downstream (F = 9.992, p = 0.010), and for rainbow trout moving upstream (F = 18.062, p = 0.002). However, no significant relationship was discovered for rainbow trout moving downstream (F = 2.259, p = 0.164).

The average daily water temperature was near 17 °C, except during the middle of the summer when it remained slightly below 15 °C for less than two weeks (Figure 5.05). Changes in the average daily water temperature was not significantly associated with the numbers of brook trout moving upstream (F = 0.015, p = 0.902) or downstream (F = 2.616, p = 0.111) during the 24-hour period between checking the weirs.

The majority of fish caught in the weirs were likely YOY brook trout and rainbow trout, based on comparisons of the fork-length estimates, projected to the fall of 1998, using the age-at-length information from the fall of 1997. The average fork lengths for brook trout and rainbow trout caught in the weirs were 6.5 and 6.0 cm, respectively (Figure 5.07(1,2)). Projecting to the fall of 1998 (September 30th), 94% of the brook trout and 81 % of the rainbow trout, were predicted to have fork lengths less than 9.0 cm (Figure 5.08(1,2)). In the fall of 1997 (September 23rd to October 7th), 89 % of the brook trout and 79% of the rainbow trout, determined to be yearlings (using scales), had fork lengths greater than 9.0 cm (Figure 5.09(1,2)). Finray aging and otolith aging corroborate with the aging data using scales (Appendix C, Dataset C-5 and C-6, respectively).

5.4 Discussion

In the first year of the study (1997), few tagged fish were re-captured within the short reaches used in 1997, and one fish for each of two reaches was caught a considerable distance outside the reach. This was contrary to the prediction of high capture frequencies of previously tagged fish based on a restricted movement hypothesis. The low catch rates suggested fish used a larger area than a few pool-riffle

Numbers Captured in the Weirs

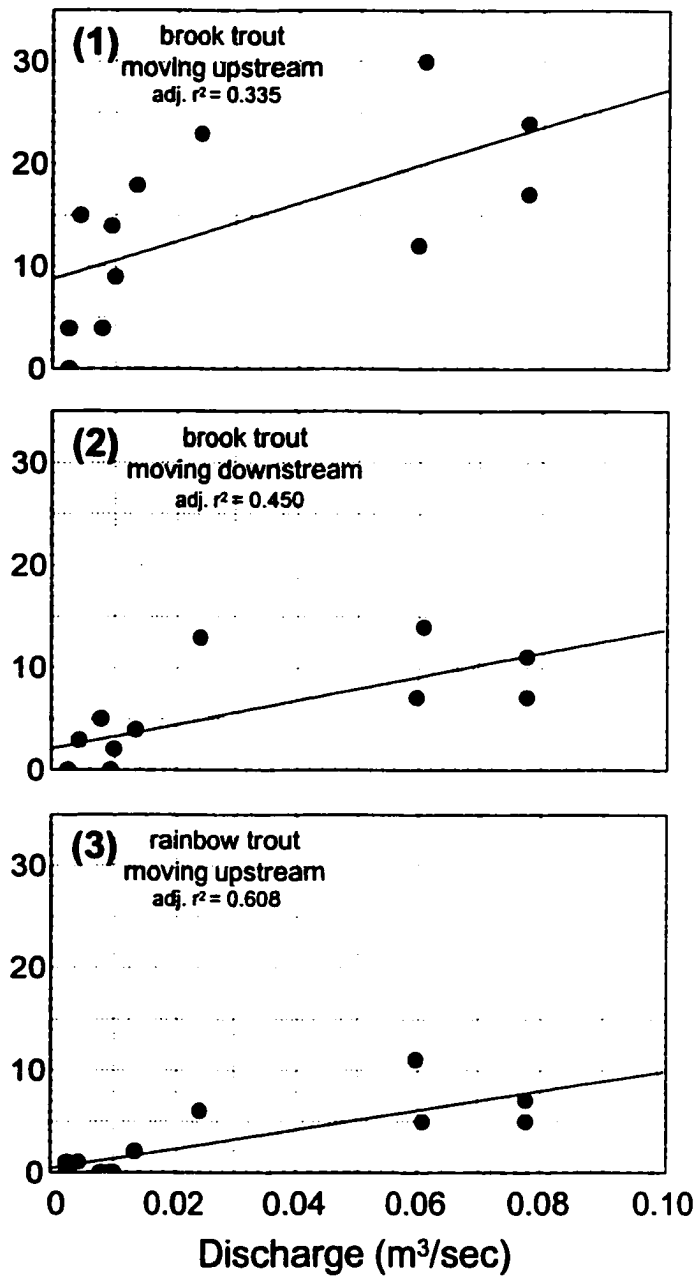

Figure 5.06: The linear relationship between stream discharge and the numbers of brook trout (1 & 2) and rainbow trout (3) captured in the weirs (both weirs combined), over the associated 48-hour period. Significant relationships were found for brook trout moving upstream (#'s captured = 8.7562 + 185.64 * discharge), downstream (# of obs. = 2.1157 + 116.12 * discharge), and rainbow trout moving upstream (# of obs. = 0.41014 + 94.579 * discharge).

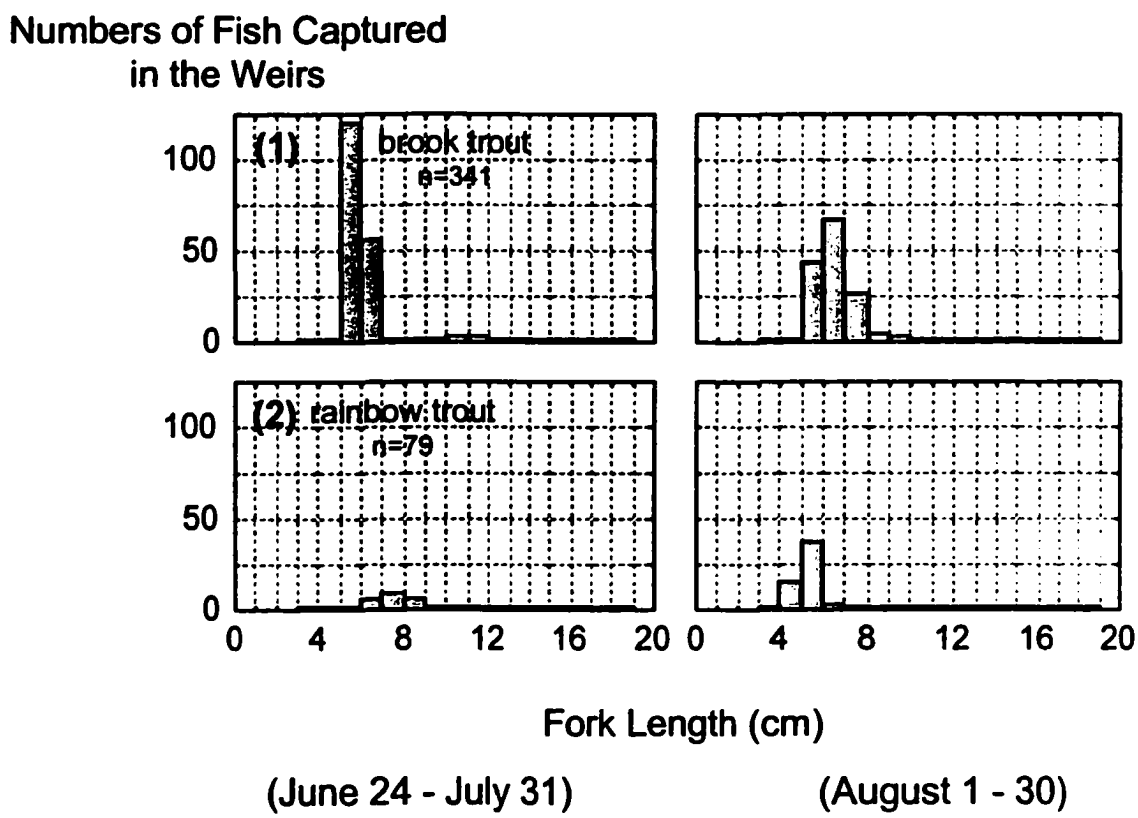

Figure 5.07: Fork lengths for individual (first-time captures) brook trout (1) and rainbow trout (2) captured in the weirs located in the Little Cypress River in 1998. The first and second half of the 67-day sampling period are displayed separately.

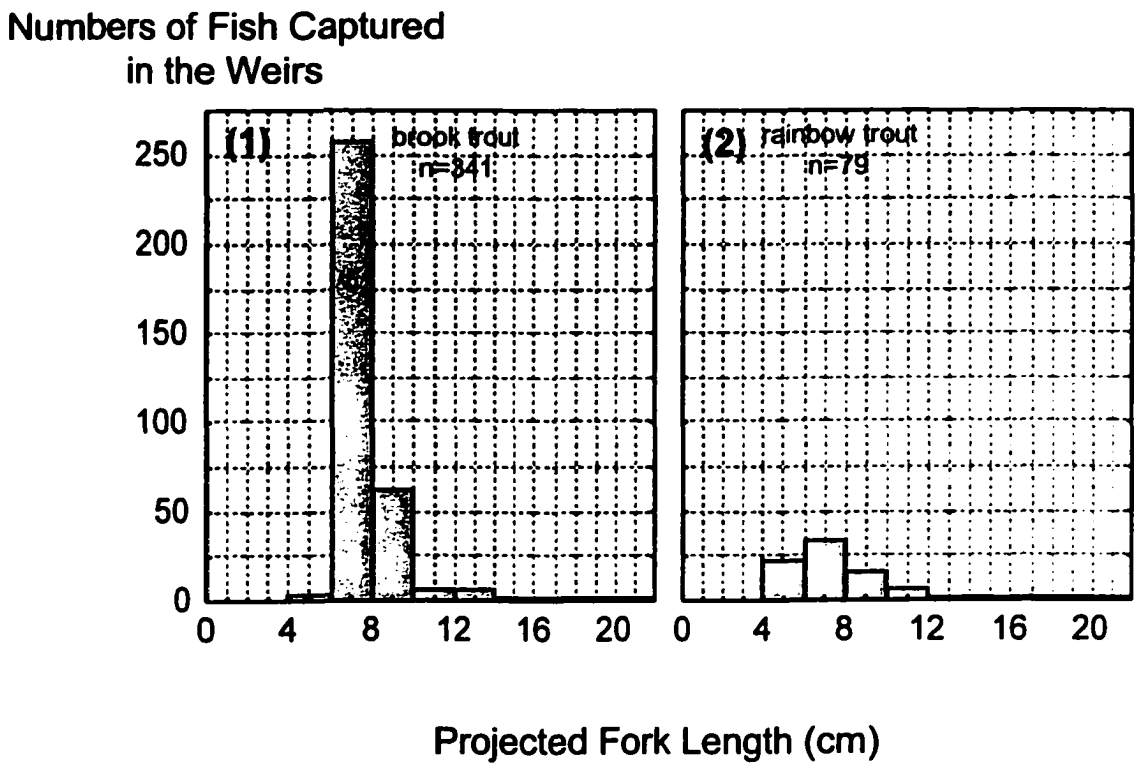

Figure 5.08: Projected fork lengths of individual (first-time captures) brook trout (1) and rainbow trout (2) captured in the weirs located in the Little Cypress River in 1998.

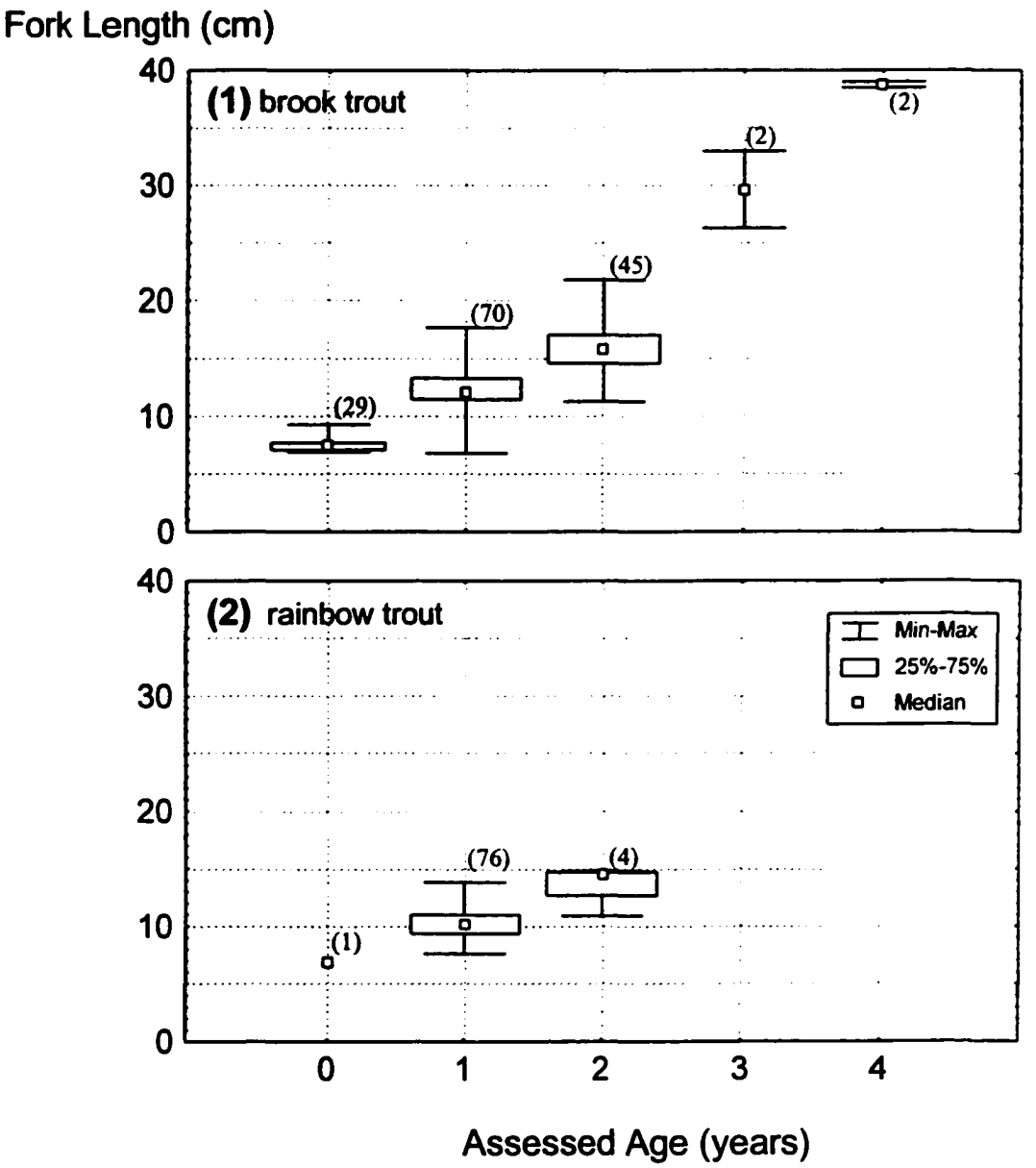

Figure 5.09: Age-at-length data for brook trout (1) and rainbow trout (2) from the three streams, using scales collected in the fall of 1997 (Sept. 23 - Oct. 7). Group sample sizes are in parentheses.

units. The conclusion that fish used a larger area was supported by catching some fish at considerable distances away. It is generally believed that the majority of stream salmonids limit their movements to a relatively small number of pool-riffle units, equalling only a few tens of metres (Northcote, 1992). However, in recent years movement by small to large fractions of stream salmonid populations are regularly confirmed through the use of radio-telemetry (Hildebrand and Kershner, 2000).

In 1997 some of the tagged brook trout and rainbow trout likely emigrated from each of the short stream reaches where they were originally tagged and released. However, determining the number of brook trout and rainbow trout that were mobile within each group requires an understanding of the associated levels of mortality and the probability of catching salmonids (catchability) in these streams using one-pass electrofishing. Stream salmonids generally experience high mortality from predation (Moyle, 1993; Gowan *et al.*, 1994). Mortality associated with predation is difficult to assess since there may be many predatory species and predation may vary for different fish species, and size or age classes of fish (Griffith, 1993). In the Au Sable River, Michigan, 79 % of age one brook trout and 45 % of age one brown trout were preyed upon by a combination of great blue heron (*Ardea herodias*), common merganser (*Mergus merganser*), otter (*Lutra canadensis*), mink (*Mustela vison*), and brown trout (Alexander, 1979). However, older brook trout and brown trout in the Au Sable River experienced 25 % and 44 % annual mortality, respectively (Alexander, 1979). In a Vancouver Island stream, common mergansers consumed 24-65 % of the potential coho salmon smolt production (Wood, 1987). Power (1980) claimed that common mergansers (*Mergus merganser*) prefer hunting in small streams and kingfishers (*Megaceryle alcyon*) residing in small headwater streams may only eat brook trout. Indeed, kingfishers and common mergansers were seen along my study streams on a few occasions, and may have contributed to the low capture rates of previously tagged

brook trout and rainbow trout..

Electrofishing, handling, and tagging can cause injuries and increase stress levels that eventually kill fish (Mitton and McDonald, 1994; Habera *et al.*, 1996). Habera *et al.* (1996) observed mortality rates averaging 9 % while testing the effects of three-pass electrofishing and handling of rainbow trout in low conductivity (< 30 uS/cm) southern Appalachian streams. Low conductivity streams require higher voltage levels to capture fish and, as a result, may cause higher mortality rates. A mortality rate of less than 10 % is comparable to other studies testing the impacts of electrofishing on salmonids in low-conductivity streams (Habera *et al.*, 1996). In my study I did not test the effects of electrofishing but I did test the mortality associated with sampling and tagging brook trout. I tested the effects of my sampling and tagging procedures on 20 small (fork lengths between 7.3 – 12.8 cm) brook trout cultured in a fish hatchery. The brook trout used in my test were placed in a raceway at the Dorion Fish Hatchery for four months of monitoring. After four months, all fish were alive and appeared in good condition. Thus, the low capture rates are not likely associated with high mortality rates related to sampling and tagging but may be partially related to mortality associated with electrofishing.

Jones and Stockwell (1995) and Kruse *et al.* (1998) found that one-pass electrofishing could be a reliable estimator of salmonid abundance in streams. Jones and Stockwell (1995) rationalized that their consistency of catching over 50 % of the salmonids on the first pass was related to the relative ease of catching salmonids, their employment of a slow, consistent electrofishing protocol, and relatively similar habitat complexity across sites. Kruse *et al.* (1998) averaged 82 % catchability while sampling trout (combinations of brook trout, cutthroat, rainbow and brown trout) in small streams of northwestern Wyoming and related their high catchability to the simple (low in structural complexity) habitat and to the low densities of salmonids. My study streams

may be low in habitat complexity. However, researchers using DC electrofishers in streams that have low habitat complexity and low conductivity may not experience high catchability since the electrofishing becomes relatively ineffective, even at high voltages (Habera *et al.*, 1996). Therefore, the probability of catching salmonids in my study streams might have been low because I used a direct current electrofisher and the water was low in conductivity.

If catchability of the salmonids was known for my study streams I would be able to estimate the number of tagged fish that vacated (died and moved) the reaches. A low catchability would mean that many fish were within the area of stream sampled but not caught. If, for example, 25 fish were captured from 75 originally tagged within an area of stream, and catchability was 100 %, then the rest of the group originally tagged (50 from 50) must have vacated (died and moved) the area. However, if catchability was 50 %, then only half of the remainder (25 from 50) must have vacated the area and the other half were within the area of stream but not caught. Therefore, I can only speculate as to whether the low numbers of tagged fish that I captured in my study are indicative of either a minority or a majority that vacated the reaches in 1997 since the catchability is not known.

In 1998, most of the captures of previously tagged brook trout (4/5) and rainbow trout (10/11) were within two pools of their original capture pool, and none emigrated from the area between the weirs. This was consistent with my prediction that most captures of previously tagged fish would occur near their original capture pool, and supports the restricted movement hypothesis. In 1997 some fish were captured farther than a few pool-riffle units from their original capture pool but I was unable to determine the degree to which far-ranging movements were being expressed. In 1998, since the previously tagged fish were originally captured in pools distributed throughout the area between the weirs, the capture locations of these previously tagged fish should be

expected to represent the degree of movement occurring beyond a few pool-riffle units. The tagged fish had the opportunity to move to any number of different pools, including those that were considerable distances from their original capture pool. However, 80 to 90 % of the brook trout and rainbow trout captures were within two pools of their original capture pool, suggesting restricted movement by the majority of fish, and movement beyond a few pool-riffle units by the minority.

During increased discharges, primarily YOY brook trout and rainbow trout moved through the weirs in an upstream direction, until the end of the summer when many also began to move downstream from the barrier. I predicted that movement would be related to some combination of size or age of fish, fish species, stream discharge, stream water temperature, and time of year (i.e. beginning, middle, or end of the summer). An explanation for the movement could be that most of the fish were moving to find territories, since YOY are the smallest, least capable of defending a territory, and are displaced when conditions are limiting (Northcote, 1992). However, the movement may also have been an innate response by some (Northcote, 1992; McCormick *et al.*, 1998). Turbulent surface water, typical of riffle areas, provides a degree of concealment for fish from predators (Scarnecchia and Bergersen, 1987). The increased discharges would have increased the areas of turbulent water in the stream and also would have made the shallow riffles of my study streams much more passable. During the upstream movement, 75 % of the fish entering into the area between the weirs were not caught further upstream at Weir 1, suggesting that they established territories or died within the reach. In many studies, fish not establishing territories eventually become displaced downstream, assumed to be a density-dependent response (Meehan and Bjornn, 1991). Similarly, downstream movement was only notably exhibited by fish located between Weir 1 and the migratory barrier. The remaining option for subordinates or colonizers, moving as far upstream as possible by the end of the

summer, would be to continue moving downstream until finding favourable conditions or die. Therefore, suitable living space may be limited in my study streams causing a displacement of subordinates when increased discharges facilitate movement to upstream locations with possibly lower levels of predation.

Based on the evidence presented, during the summer, most of the larger brook trout and rainbow trout in these streams use a restricted area (a few pool-riffle units), and smaller brook trout and rainbow trout may move hundreds of metres. In this section of the study, I was testing the hypothesis that brook trout and rainbow trout in these streams are restricted in their movements. If physical habitat or food was known to be limiting in these streams beforehand, restricted movement by larger salmonids and increased mobility by smaller salmonids would have been logical predictions to make. However, further research needs to be conducted to validate these conclusions. Information regarding the catchability and mortality of salmonids during the summer in these streams would be beneficial for any future study of this type, and further interpretation of these results.

Chapter 6: General Discussion

Brook trout and rainbow trout use small tributary streams along the north shore of Lake Superior to fulfill part or all of the stages of their life cycles. These streams are located on the Canadian Shield and contain some of the most infertile water where salmonid populations are found (O'Connor and Power, 1976). Below migratory barriers, the streams provide rearing habitat for young salmonids, but some brook trout may also remain for their entire lives (Power, 1980). Above migratory barriers, isolated brook trout populations have survived for thousands of years (Northcote, 1992). Short summers and low water velocities put added constraints on the ability of these salmonids to meet their energetic requirements during the summer. Although their behaviour in these streams does not appear to be unprecedented, the results of this study attest to the ability of salmonids to survive under extreme conditions.

It was predicted that brook trout would be positively associated with physical habitat structure in streams. However, the results of this study do not strongly support this prediction. Using baited minnow traps, brook trout were captured more often in pools with structure as opposed to no structure, but when structure was added to pools that were previously devoid of structure there was no significant increase in brook trout numbers. Also, brook trout biomass for entire pools was not strongly associated with structure in pools. The lack of a clear association is possibly due to an over-abundance of habitat structure (i.e. not a limiting factor) in these streams.

Initially, low capture rates during this study suggested the use of much larger areas of stream by the brook trout and rainbow trout but most capture locations of tagged brook trout and rainbow trout were less than three pools from their original capture pool. It was initially assumed that catchability of fish would be high because of the low number of fish inhabiting my study streams and the relatively unrestricting habitat. However, catchability may not be high since similar low capture rates occurred during the second year of the study and yet it was clear that none of the tagged fish had left the reach.

Mortality rates associated with capture and sampling methods were assumed to be low over the period of the study since no mortality was associated with my tagging experiment and only one-pass electrofishing was used, as opposed to multiple-pass electrofishing, to reduce associated mortality. However, higher mortality rates than expected could partially account for my inability to capture tagged fish within the reaches since nothing is known about the mortality rates that may be associated with predation and natural causes.

If most of the brook trout and rainbow trout in these streams remain within an area of only a few consecutively located pools for up to eight weeks, some feature of the area of stream must have been beneficial in providing the necessities for survival. Stream salmonids are usually territorial, meaning they defend a foraging area (Fausch and White, 1981; Biro *et al.*, 1997; Gowan *et al.*, 1994; Chapman, 1966). Typically, as pressure increases on the stream's resources, of available food and space, stream salmonids become more aggressive and territories increase in size (Murphy and Meehan, 1991; Marcus *et*

al., 1990; Bjornn and Reiser, 1991; Grant and Kramer, 1990). Biro *et al.* (1997) found that territorial brook trout in streams became non-territorial, active foragers when inhabiting a lake which might be similar to the response of brook trout and rainbow trout in streams that have very low velocities during the summer. Therefore, if conditions in my study streams were food limited then the use of a few consecutively located pools or more may have been a necessary response by many of the brook trout and rainbow trout to meet their energetic requirements.

Knowledge of the catchability and mortality rates of the brook trout and rainbow trout in my study streams would have enhanced my ability to interpret the findings. Due to this lack of knowledge, tagged fish that were not re-captured within the reaches may have moved, avoided capture, or died. In past studies of movement by stream salmonids, catchability and mortality were often assumed to explain the fate of the fish that were never captured again (Gowan *et al.*, 1994). However, in recent years movement by stream salmonids has been regularly confirmed through the use of radio telemetry (Hildebrand and Kershner, 2000). Therefore, further investigations are necessary to substantiate my conclusions since they are based on assumptions of low or high mortality for brook trout and rainbow trout, depending on their level of catchability in my study streams.

Further limitations to the conclusions based on my findings involve the length of reach that was used during the second year of the study and the time that was allowed to pass before returning to sample the fish within the reach. No

tagged fish exited the reach but the reach was long (466 m) and sampling for tagged fish occurred after a considerable length of time (one month) had passed. Therefore, tagged fish could have moved hundreds of metres during the month without detection, yet it was assumed that the locations of capture, with respect to previous capture pools, were indicative of the length of stream required by each fish. Hildebrand and Kershner (2000), using radio-telemetry to study the movement of cutthroat trout in Beaver Creek, Idaho-Utah, found that a change in temporal scale could completely alter the interpretation of their results. For example, movement data analysed by Hildebrand and Kershner (2000) using locations recorded during diel monitoring indicated that cutthroat trout in their study streams were mobile. However, when Hildebrand and Kershner (2000) analysed the results from weekly positions, the results were more indicative of sedentary behaviour. Similarly, the methods used during the second year of my study may have been ineffective at detecting the general length of stream used by the brook trout and rainbow trout in my study streams, since the reach was long and only sampled once after a considerable length of time had passed.

The problems associated with determining the habitat preferences of brook trout and rainbow trout in my study streams could have been alleviated by direct observations in the natural stream environment. Observational studies of habitat use by stream salmonids have been conducted on many occasions and leave little doubt in interpretation of the results. McLaughlin *et al.* (1994) successfully used bank-side observational methods to study the water column use and diet of recently emerged brook trout in Ontario streams. Similarly,

Jenkins (1969) gave valuable information regarding the habitat use and territorial behaviour of brown and rainbow trout in mountain streams of California. Fausch and White (1981) used observational methods, while snorkelling within the stream, to examine habitat use by brook trout. Under infertile conditions when statistical relationships between fish biomass and habitat features may be weak, observational methods might be the most prudent use of time and money to determine habitat use by stream salmonids.

Future research of brook trout and rainbow trout movement patterns in small streams might benefit with the use of two-way weirs placed at the top and bottom of shorter, consecutively located stream reaches. The length of the reach in relation to the extent of the movements by the fish under study is a critical aspect of an experimental design to detect movement (Gowan *et al.*, 1994). The use of short reaches would negate the need to electrofish the area but would require sampling the fish, captured in the weirs, on a daily basis. Low numbers of larger fish captured in weirs placed at the top and bottom end of short stream reaches would reveal limited movement by the stream salmonids within the stream under study. However, movements by stream salmonids beyond a small area would still be detected using consecutively located short reaches since they would collectively encompass a large area. Daily sampling would also result in an understanding of the length of stream used by stream salmonids during different conditions and time of year. In theory, the experimental design would enable the detection of most movement while reducing the mortality associated

with electrofishing and alleviating the problems asociated with potentially low catchability.

Land-use activities, such as forest harvesting, might affect the characteristics or habitat features of streams that are important to salmonids, but if these features are not previously identified it could be difficult to make appropriate management decisions to sustain healthy populations of the principal species. Although habitat structure may not be as important as food, more research is necessary to determine the appropriate methods and spatial scale required to clearly identify the habitat use preferences of the brook trout and rainbow trout in these streams. At a minimum, this study provides baseline information regarding habitat and species composition of these streams that could benefit future management decisions if they are ever significantly disturbed by land-use activities.

If the few remaining populations of coasters in Lake Superior are to be protected and if the plans to rehabilitate coasters in areas where they once thrived are to be successful, then the time to learn more about the role of small tributary streams in their life cycles and their behaviour amongst other, introduced salmonids within these streams is imminent. Anecdotal evidence has identified tributaries of Lake Superior as important spawning and rearing grounds for coaster brook trout, and concerns over the impact of introduced salmonids on coasters has been an immutable concern. The size distribution of brook trout caught in these streams indicates that mainly younger fish use these streams, therefore, tributaries of Lake Superior may be a necessary component of the

rearing habitat for coasters. Additionally, habitat use preferences by brook trout appeared to be affected by the presence of rainbow trout, relative to the above barrier group. Furthermore, stream rehabilitation methods should now begin to include considerations for smaller salmonids that may aspire to move upstream during low flow conditions following the spring runoff. The information gathered during this study will likely contribute, to some degree, in alleviating the unfortunate state of coaster brook trout in Lake Superior, but we are still a long way from meeting the tremendous challenge of that worthy and fundamental goal.

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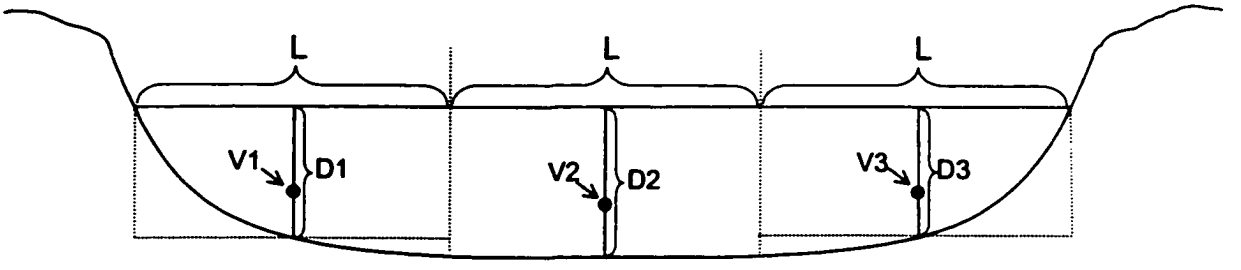
APPENDIX A: FORMULAE



V1, V2 or V3 = velocity locations (V1, V2, V3, etc) at 60 % of the depth

D1, D2 or D3 = depth locations taken at halfway point along the increment

Stream Discharge = (L x V1 x D1) + (L x V2 x D2) + (L x V3 x D3)

Formula A-1: The formula, and accompanying diagram, demonstrate the procedure and calculations involved in determining stream discharges. The diagram is a simplified example, used for clarification.

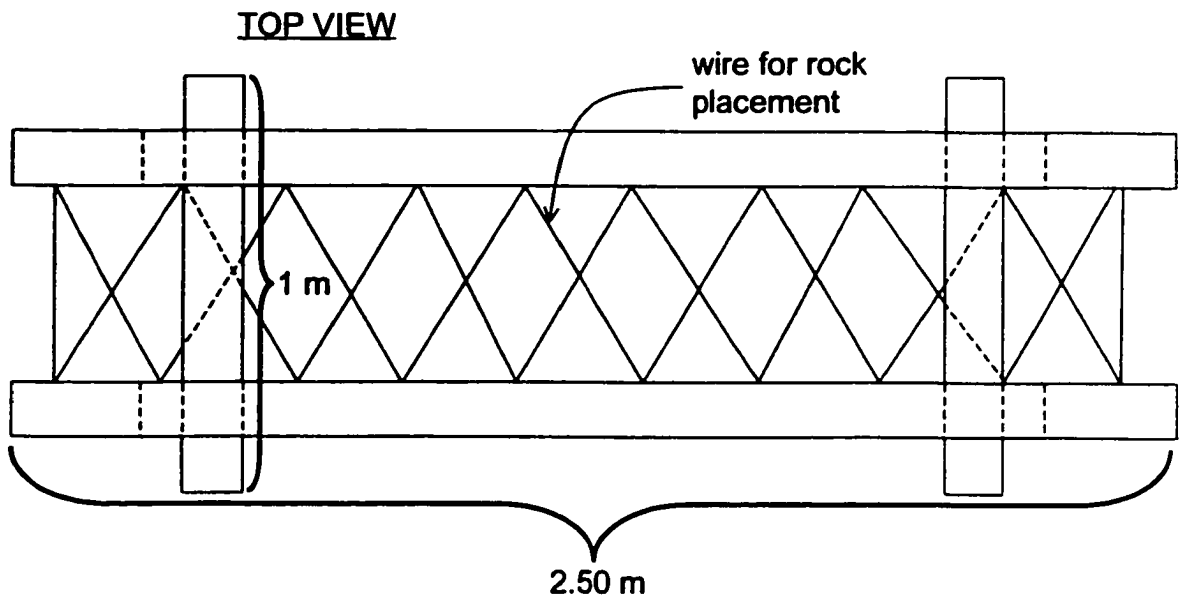
$$\text{Volume of Submerged Wood} = \text{length} \times 0.75 \times \text{average diameter}^2$$

Formula A-2: The formula used to calculate the volume of submerged wood found within pools. This included wood with an average diameter (roughly estimated in the field) of five centimetres or greater.

$$\text{FL forward-calculated} = \text{FL for first capture date} + \left[\text{days betw. first capture and Sept. 30} \times \text{estimated growth/day}\right]$$

Formula A-3: A forward-calculation formula used to make a projection or estimate of the fork length that a particular fish caught in the weirs would have reached by September 30, 1998. This formula is based on an assumed daily growth rate of 0.20 mm/day.

APPENDIX B: DIAGRAMS



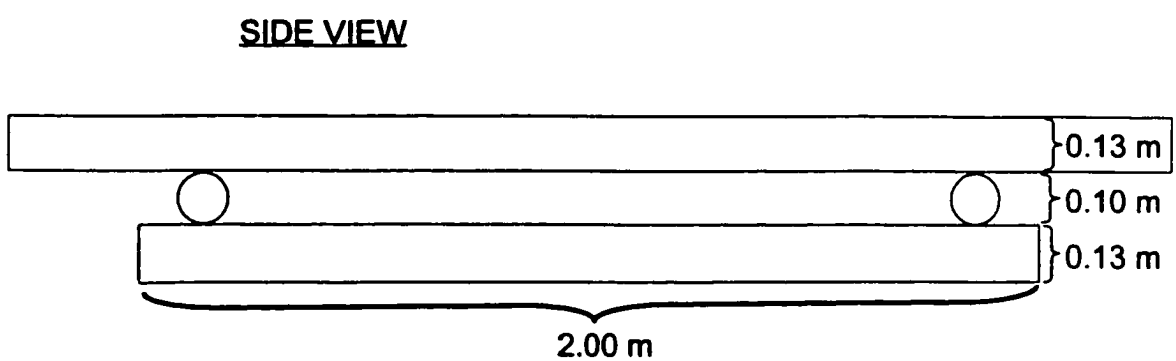


Diagram B-1: Top and side views of the habitat structures. Not depicted here, are the cedar branches that were wired to the top of the structures to provide direct-shade, along with the rocks that held the structures down and provided further structure.

APPENDIX B: DIAGRAMS (continued)

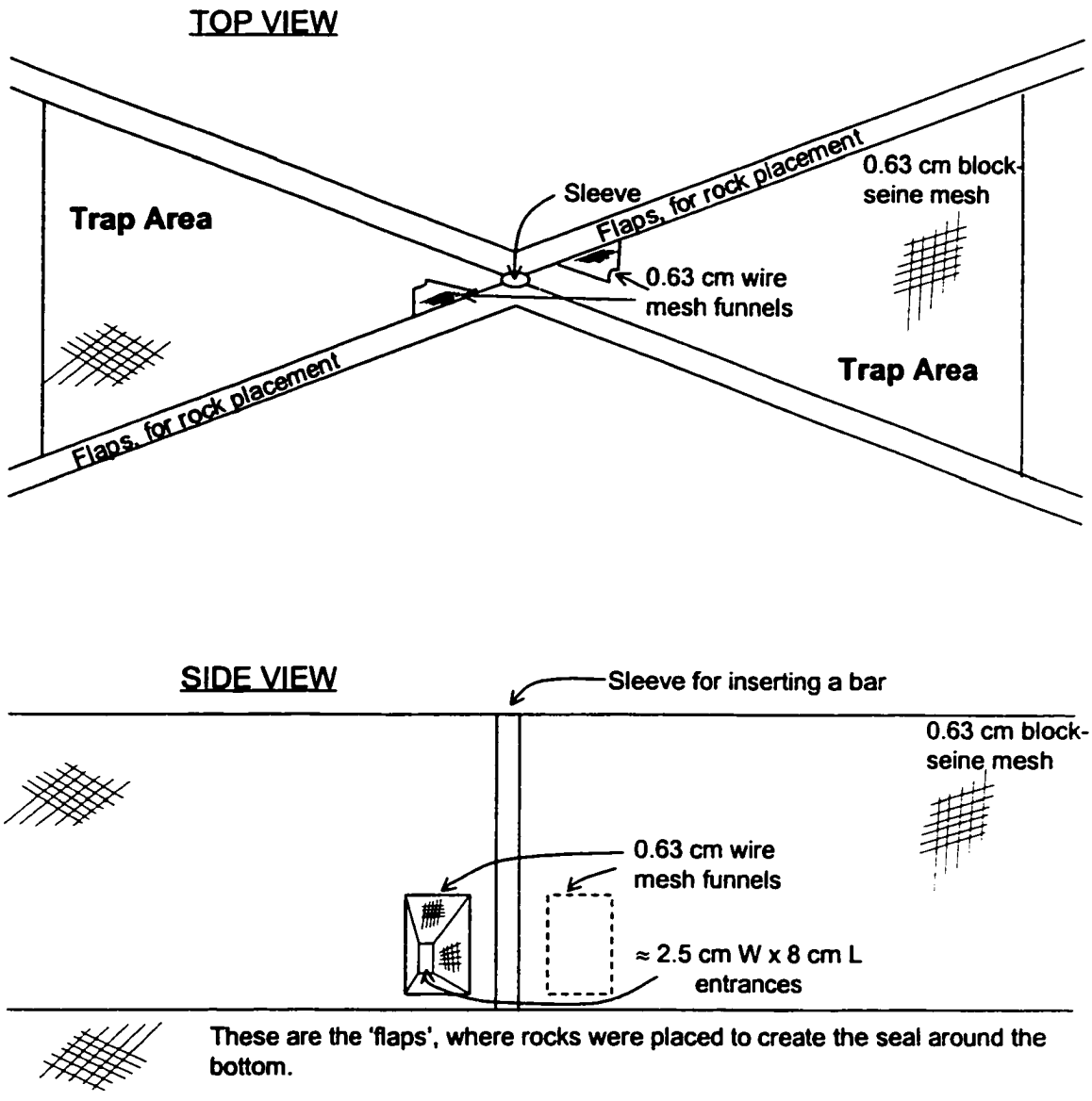


Diagram B-2: Top and side views of the weirs that were constructed for this study. Not depicted here, are the ropes that were needed across the stream to give the structure support from above. The rocks placed on the 'flaps' and the metal bar inserted through the 'sleeve' (and driven into the stream bottom), hold the structure rigid within the stream.

APPENDIX C: DATASETS

Dataset C-1: Data from the experiment conducted at the Dorion Fish Hatchery, to assess mortality rates associated with the use of PIT tags on small brook trout.

Fish #	Tag #	Fork Length (cm) on May 28, 1997	Fork Length (cm) on October 3, 1997	Alive?
1	411347562A	12.4	17.8	Yes
2	41125D0D44	7.7	10.5	Yes
3	41133F016D	9.8	16.0	Yes
4	4113494F0D	10.4	16.7	Yes
5	41130E193A	8.3	13.3	Yes
6	411263519E	10.7	16.2	Yes
7	4112786B1F	9.0	15.1	Yes
8	4113167B4F	8.2	12.9	Yes
9	4113505442	9.3	15.3	Yes
10	41132F2D5E	10.6	16.3	Yes
11	41127E6728	10.1	15.4	Yes
12	4113187216	11.6	16.8	Yes
13	41133A2A27	12.5	19.8	Yes
14	41131F034E	11.4	18.5	Yes
15	4113252A63	12.5	19.9	Yes
16	4113615541	9.8	16.2	Yes
17	41126B033C	9.7	15.2	Yes
18	41125F2D20	9.1	14.2	Yes
19	411276523B	12.8	18.6	Yes
20	41127A0449	7.3	10.3	Yes

APPENDIX C: DATASETS (continued)

Dataset C-2: Weight estimates that were used to determine the total pool biomass values for the pools used in the principal component analysis above barriers, in 1998. These fish escaped being caught, but length estimates were used to form the weight estimates so that the weights of these fish would not be excluded from the total pool biomass values.

Stream	Pool	Species	Fork Length Estimate (cm)	Weight Estimate (g)	Pool Biomass (g/pool)
Dublin Upper	3	BT	15.5	25	166
Dublin Upper	7	BT	10.5	10	212
Dublin Upper	7	BT	12.5	15	212
Dublin Upper	14	BT	8.5	6	367
Dublin Upper	14	BT	11.5	12	367
Dublin Upper	15	BT	9.5	10	115
Dublin Upper	16	BT	9.5	10	114
Dublin Upper	16	BT	9.5	10	114
Dublin Upper	30	BT	11.5	12	280
Dublin Upper	31	BT	10.5	10	122
Dublin Upper	36	BT	12.5	15	17
Dublin Upper	45	BT	10.5	10	351
L. Cypress Upper	3	BT	8.5	6	95
L. Cypress Upper	3	BT	8.5	6	95
L. Cypress Upper	9	BT	11.5	12	31
L. Cypress Upper	11	BT	8.5	6	117
L. Cypress Upper	11	BT	12.5	15	117
L. Cypress Upper	12	BT	9.5	10	91
L. Cypress Upper	12	BT	9.5	10	91
L. Cypress Upper	13	BT	9.5	10	54
L. Cypress Upper	13	BT	11.5	12	54

APPENDIX C: DATASETS (continued)

Dataset C-3: Weight estimates that were used to determine the total pool biomass values for the pools used in the principal component analysis below barriers, in 1998. These fish escaped being caught, but length estimates were used to form the weight estimates so that the weights of these fish would not be excluded from the total pool biomass values.

Stream	Pool	Species	Fork Length Estimate (cm)	Weight Estimate (g)	Total Pool Biomass (g/pool)
Dublin Lower	-	-	-	-	-
L. Cyp. Lower	-	-	-	-	-
MacInnes	16	BT	12.5	15	876
MacInnes	21	BT	11.5	12	32
MacInnes	33	BT	8.5	6	28
MacInnes	35	BT	6.5	2	55
MacInnes	28	RT	12.5	15	74
MacInnes	31	RT	8.5	6	41
MacInnes	39	RT	9.5	10	94
MacInnes	39	RT	10.5	10	94
MacInnes	40	RT	10.5	10	61

Dataset C-4: Weight estimates that were used to determine the total pool biomass values for the pools of the habitat manipulation experiments, in 1998. These fish escaped being caught, but length estimates were used to form the weight estimates so that the weights of these fish would not be excluded from the total pool biomass values.

Stream	Before or After Habitat Additions	Pool	Species	Fork Length Estimate (cm)	Weight Estimate (g)	Total Pool Biomass (g/pool)
Dublin Upper	Before	3	BT	11.5	15	271
Dublin Upper	Before	9	BT	10.5	10	332
Dublin Upper	After	1	BT	7.5	5	88
Dublin Upper	After	1	BT	12.5	15	88
Dublin Upper	After	1	BT	12.5	15	88
Dublin Upper	After	4	BT	9.5	9	109
Dublin Upper	After	5	BT	7.5	5	166
Dublin Upper	After	7	BT	9.5	9	224
Dublin Upper	After	9	BT	7.5	5	300
Dublin Upper	After	9	BT	17.5	55	300
MacInnes	Before	1	BT	12.5	15	866
MacInnes	Before	7	BT	8.5	6	22
MacInnes	Before	10	RT	10.5	10	61
MacInnes	After	-	-	-	-	-

APPENDIX C: DATASETS (continued)

Dataset C-5: Assessed ages and associated fork lengths, capture locations, and dates of capture for brook trout aged using finrays. Aging assessments were performed by Susan Mann of the Northwest Science and Information section of the Ontario Ministry of Natural Resources.

Stream Section	Date	Fork Length (cm)	Assessed Age (yrs)
Little Cypress Lower	23-Aug-98	6.7	0
Little Cypress Lower	24-Sep-97	7.4	0
Little Cypress Lower	22-Aug-98	7.5	0
Little Cypress Lower	25-Aug-98	7.9	0
Little Cypress Lower	26-Aug-98	8.8	0
MacInnes	30-Sep-97	8.8	0
MacInnes	30-Sep-97	10.2	0
Dublin Lower	23-Sep-97	11.3	0
Little Cypress Lower	24-Sep-97	11.4	1
Little Cypress Lower	24-Sep-97	12.0	0
MacInnes	30-sep-97	12.4	0
Little Cypress Lower	24-Sep-97	13.8	1
Little Cypress Lower	24-Sep-97	14.1	1
Dublin Lower	23-Sep-97	14.1	1
Dublin Lower	23-Sep-97	16.6	1
Dublin Lower	23-Sep-97	17.0	2
Little Cypress Lower	24-Sep-97	18.1	1
Little Cypress Lower	24-Sep-97	19.5	2
Dublin Lower	23-Sep-97	21.8	3
Dublin Lower	23-Sep-97	23.5	2
Dublin Lower	23-Sep-97	26.3	2
Little Cypress Lower	24-Sep-97	28.7	4
MacInnes	30-Sep-97	30.0	2
Little Cypress Lower	24-Sep-97	33.0	3
Dublin Lower	23-Sep-97	38.5	3
Little Cypress Lower	24-Sep-97	39.0	3

APPENDIX C: DATASETS (continued)

Dataset C-6: Assessed ages and associated fork lengths, capture locations, and dates of capture for brook trout aged using otoliths. Aging assessments were performed by John Casselman of the Aquatic Ecosystems section of the Ontario Ministry of Natural Resources.

Stream Section	Date	Fork Length (cm)	Assessed Age (yrs)
Dublin Upper	10-Jul-97	5.0	0
MacInnes	13-Jul-98	5.6	0
Little Cypress Lower	21-Jul-98	6.5	0
Little Cypress Lower	21-Jul-98	6.5	0
Little Cypress Lower	25-Jun-98	6.7	0
Dublin Upper	19-Aug-97	9.8	2
Little Cypress Lower	25-Jun-98	10.1	1
MacInnes	13-Jul-98	10.2	1
MacInnes	14-Jul-98	10.8	1
Dublin Upper	7-Jul-98	11.0	2
Little Cypress Lower	25-Jun-98	11.1	2
Dublin Upper	10-Jul-97	13.4	2
Dublin Upper	7-Aug-97	18.2	3